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⊕EPA # Research and Development

REVIEW OF EMISSION FACTORS AND
METHODOLOGIES TO ESTIMATE
AMMONIA EMISSIONS FROM ANIMAL
WASTE HANDLING

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Prepared by

National Risk Management
Research Laboratory
Research Triangle Park, NC 27711

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E. Timothy Oppelt, Director
National Risk Management Research Laboratory

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REVIEW OF EMISSION FACTORS AND METHODOLOGIES TO ESTIMATE AMMONIA EMISSIONS FROM ANIMAL WASTE HANDLING

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ABSTRACT

Currently, approximately 80% of ammonia (NH_3) emissions in the United States (U.S.) originate from livestock waste. This report summarizes and discusses recent available U.S. and European information on NH_3 emissions from swine farms and assesses the applicability for general use in the U.S., and North Carolina in particular. In addition, limited information on NH_3 emissions from other farm animals is included, as well as some information on methane emissions from anaerobic animal waste lagoons and nitrous oxide emissions from swine waste spray fields. The report discusses a comprehensive mass balance approach that may be used to estimate emissions for certain livestock and poultry operations.

The emission rates for the houses calculated by various methods show good agreement and suggest that the houses are a more significant source than previously thought. It is believed that there is enough basis to recommend a general emission factor for houses of 3.7 ± 1.0 kg NH_3/year/finisher pig or 59 ± 10 g NH_3/kg live weight/year. This value is supported by the 4.3 kg NH_3/year/finisher pig reported for several pull-plug houses which were tested in the summer of 2000. For lagoons, it was found that there is good similarity between the field test results and the number calculated by a mass balance method. The suggested annual NH_3 emission factor based on field tests at one swine farm lagoon in North Carolina is 2.4 kg/year/pig. The emission factor for lagoons is based on field tests at only one lagoon and is considered to be less accurate than the emission factor for houses. Emission rates from spray fields were estimated using a total mass balance approach, while subtracting the house and lagoon emissions.

The total emission rates for finishing pigs at the test farm compared well to the total rate established by a mass balance approach based on nitrogen intake and volatilization. Therefore, it was concluded that a mass balance approach can be helpful in estimating NH_3 emissions from swine farms. Assuming that the swine population at the test farm was a self-sustaining population, similar to the average swine population in North Carolina, a general emission factor of 7 kg NH_3/pig/year was developed. This emission factor is comparable to three general European emission factors, which varied from 5 to 6 kg NH_3/pig/year.

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ABBREVIATIONS AND ACRONYMS

AAP	Average animal present
APPCD	USEPA's Office of Research and Development, Air Pollution Prevention and Control Division
°C	degrees Celsius
cm	centimeter
g	gram
Gg	gigagram (10^9 grams)
kg	kilogram
kW	kilowatt
mg	milligram (10^{-3} grams)
m^2	square meter
N	nitrogen
NCDENR	North Carolina Division of Environment and Natural Resources
NH_3	ammonia
NH_4^+	ammonium
OP-FTIR	open-path Fourier transform infrared spectroscopy
pH	minus logarithm of hydrogen ion concentration
Tg	teragram (10^{12} grams)
TKN	Total Kjeldahl Nitrogen
U.S.	United States
USDA	U.S. Department of Agriculture
USEPA	U.S. Environmental Protection Agency

1   INTRODUCTION

In the United States (U.S.) the atmospheric deposition of ammonia (NH_3) and other nitrogen compounds has received renewed attention as a major route of entry into watersheds, especially the lower river basins and coastal estuaries of the eastern U.S. These nitrogen compounds, particularly the reduced forms such as NH_3, are available as plant nutrients and add to the eutrophication problems already of concern in these coastal areas (USEPA, 1997). Atmospheric NH_3 further contributes to the formation of fine particulate matter (aerosols) by reacting with acid gases from combustion sources (Harris, 2001; Aneja et al., 2000).

Animal waste management is the biggest source of NH_3 emissions.[1] A report by Battye et al. (1994) concluded that about 80% of NH_3 emissions in the U.S. originates from livestock waste. Also according to *Atmospheric Emission Inventory Guidebook* (AEIG, 1998) of the European Environment Agency, over 80% of total NH_3 emissions in Europe originate from animal husbandry. On a global scale, the fraction of NH_3 from domestic animals was estimated to be about 50%. The difference results from alternate waste management techniques and from the nitrogen-rich diet that is fed to domestic animals in Europe and other countries with industrialized livestock and poultry operations (Bouwman et al., 1997; Schlesinger and Hartley, 1992).

As in northern Europe, an increasing tendency towards industrialization of farming practices in the U.S. over the last decade and a half has resulted in increased farm size and confinement of animals. Certain geographic shifts have further intensified potential environmental effects from NH_3. For example, in the last 6 years, Eastern North Carolina, a sparsely populated agricultural region characterized by sandy soils and flat terrain, has been the scene of explosive growth in intensive livestock production facilities, principally swine[2] and poultry. In 1991, the average swine population was about 4.5 million. By 1995, the number had increased to 8.2 million, to reach about 10 million in 1997. Due to a moratorium imposed by the Legislature of North Carolina in 1997, the swine population has remained at around 10 million. (Note that these numbers reflect populations at a given time. Annual production numbers are about twice as high, because it takes half a year to raise swine up to market weight.) Swine facilities in North Carolina have primarily

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[1] In this report, emissions are understood as air emissions, unless otherwise indicated.

[2] Hogs and pigs constitute the majority of the swine family, which also includes wild boars. Pigs are young swine; whereas, hogs are fully grown swine. Farrows are piglets. Sows are female hogs. The terms hog, pig, and swine are often used interchangeably.

located in the watersheds of the coastal plain, with the highest concentrations in the Neuse and Cape Fear watersheds.

Poultry production in North Carolina has also increased over the last two decades, the two most important categories being broilers and turkeys. Statewide, approximately 777 million broilers and 54 million turkeys were produced in 1999. Most broilers are being produced in the western and south central parts of the State, but most turkeys are being raised in the southeast district, where the concentration of hog farms is also highest (Sheldon, 2001).

In a 1997 report to Congress, the U.S. Environmental Protection Agency (USEPA) estimated that 27% of the annual nitrogen concentrations in the Chesapeake Bay result from atmospheric deposition. In North Carolina's Albemarle-Pamlico Sound, the proportion was estimated at 44%. As a result of increased public awareness concerning NH_3 and its impact on the environment, North Carolina has become a center for atmospheric NH_3 emission and deposition research in the last few years. In 1996 the Division of Air Quality of the North Carolina Department of Environment and Natural Resources began a program to evaluate NH_3 emissions, deposition, and subsequent environmental impacts from deposition. The primary purpose of this program was to conduct modeling, but it soon became evident that a lot of emission information was needed before any modeling could be accomplished. During the next few years, the State coordinated a significant test effort to determine NH_3 emission factors from swine farms. It was decided to initially focus on lagoons, as they were believed to be the major source of NH_3. These efforts are detailed in the proceedings of two workshops that were held in Raleigh, North Carolina, in March 1997 (NCDENR, 1997) and in Chapel Hill in June 1999 (NCDENR, 1999). Field tests and field test results are summarized in Chapter 3 of this report.

1.1   PURPOSE OF THIS REPORT

This report summarizes and discusses recent available U.S. and European information on NH_3 emissions from swine farms and assesses the applicability for general use in the U.S., and North Carolina in particular. In addition, limited information on NH_3 emissions from farm animals other than swine is included, as well as some information on methane emissions from anaerobic animal waste lagoons and nitrous oxide emissions from swine waste spray fields. The temporal cut-off is 1994, because earlier comparable information is assumed to have been detailed in the Battye report, which was published in 1994. Because no comprehensive U.S. NH_3 emissions methodology was found, this report includes a detailed review of the most common European methodology, which is based on a mass balance approach. Furthermore, the report highlights the difference between the European and the U.S. conditions (especially for North Carolina) and

suggests how European information can be used to assist in developing emission factors for U.S. emissions from houses[3], waste storage, and land application. Data gaps are defined that impede the development and application of improved emission factors in the U.S. The main focus of this report is on NH_3 emissions from swine operations, because the best and most recent available U.S. and European NH_3 emissions information is on swine farms. The report does not address deposition or control technologies or practices.

Chapter 1 provides an introduction to the NH_3 issue, as well as some background information on NH_3 chemistry and emissions pathways from animal waste. Further supporting information is provided in Appendix A which addresses general farming and animal waste practices in the U.S. Chapter 2 summarizes findings from European and general American literature pertaining to NH_3 emissions, whereas Chapter 3 focuses on the field tests that were conducted in the southeastern U.S. Chapter 4 introduces a mass balance approach based on European models, while the available emissions data are applied and compared in Chapter 5. Appendix B provides a review of the information that was collected on methane and nitrous oxide emissions associated with animal waste. A synopsis of field test methods that were used to sample NH_3 air emissions is included in Appendix C.

1.2   BACKGROUND INFORMATION ON AMMONIA AND AMMONIA EMISSIONS FROM ANIMAL WASTE

Ammonia is a compound of nitrogen and hydrogen. Chemically it is a base, and it reacts in the atmosphere with acidic species to form ammonium (NH_4^+) sulfate, NH_4^+ nitrate, or NH_4^+ chloride. Deposition of these salts has been identified as a major cause of soil acidification in The Netherlands because oxidation of NH_4^+ via nitrifying bacteria releases two H^+ ions into the soil (Aneja et al., 2000). An excess of available nitrogen further leads to eutrophication of surface water and soil. Nitrogen is the limiting factor in many ecosystems. If additional nitrogen is supplied, it is initially turned into rapid growth of the stem, trunk, and foliage of the plant. Plants experiencing this growth characteristic become more susceptible to insects and fungal infections. The root system remains small in proportion to the foliage, making the plant or tree more sensitive to drought and frost. Furthermore, in a nitrogen-limited ecosystem that is receiving excess nitrogen, native plants will be replaced by invasive, nitrogen-loving species, such as nettles, grasses, and brambles (IKC/RIVM, 1995).

Ammonium ions from animal waste sources are formed as the result of the microbial breakdown of urea by the enzyme urease. For mammals, this process starts when the urea in the voided urine comes in contact

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[3] The barns where the animals are kept are also called houses. These terms are used interchangeably throughout the report.

with the enzyme urease that is present in the excreted feces. For birds, waste protein products are excreted by the kidneys as uric acid, which can be broken down to urea. The enzyme urease is also active in the soil where residual urea and other nitrogenous compounds are broken down. In aqueous solutions, there exists an equilibrium between NH_3 and NH_4^+. The ratio of dissolved NH_3 to total ammoniacal nitrogen ($NH_3 + NH_4^+$) in the solution increases with increasing pH and temperature. The dissolved NH_3 may then volatilize to the surrounding air (Bouwman et al., 1997).

To define NH_3 emissions, the animal waste source category is usually subdivided into four sub-categories: animal houses, waste storage, land application of the waste, and pasture use. The pasture (free range) category is pertinent for beef and dairy cows because these may spend all or part of their lives in pasture. Also, other minor categories, such as sheep and goats are typically kept in pasture. Per quantity of waste, NH_3 emissions from pasture will be less, compared to collected waste. The main reason is that the waste (solids and urine) is spread out more and can, therefore, better be absorbed by the ground or by vegetation, especially after precipitation events. Swine and poultry are typically entirely confined in houses. A modern swine farm may have up to 10 houses with some 800 pigs per house. In the case of a chicken farm, one house may contain up to a 100,000 chickens. Waste storage systems include lagoons, storage pits, or the lower part of a house itself in the case of chickens. Effluent from the lagoon or manure from a storage pit or the house is typically land applied, where it is distributed over a field. Because this study looks at confined animal farming operations, emissions from pasture are not further discussed. Figure 1 provides an overview of likely NH_3 emission pathways from farm animal waste. Appendix A includes descriptions of general animal farming and animal waste handling techniques in the U.S.

Ammonia emissions from farm animal waste depend on the average nitrogen content in the waste for different animal categories and subsequent NH_3 losses during housing, storage/treatment of wastes outside the building, and application of the animal waste to the land. These three subcategories should be viewed as one system. For example, a change in animal diet will affect emissions from excretions in the house, but subsequently also emissions from the waste storage site (e.g., the lagoon), and ultimately those from land application of the lagoon effluent or solids. The interdependence is illustrated further by the many factors that have been shown to influence NH_3 emissions from livestock waste.
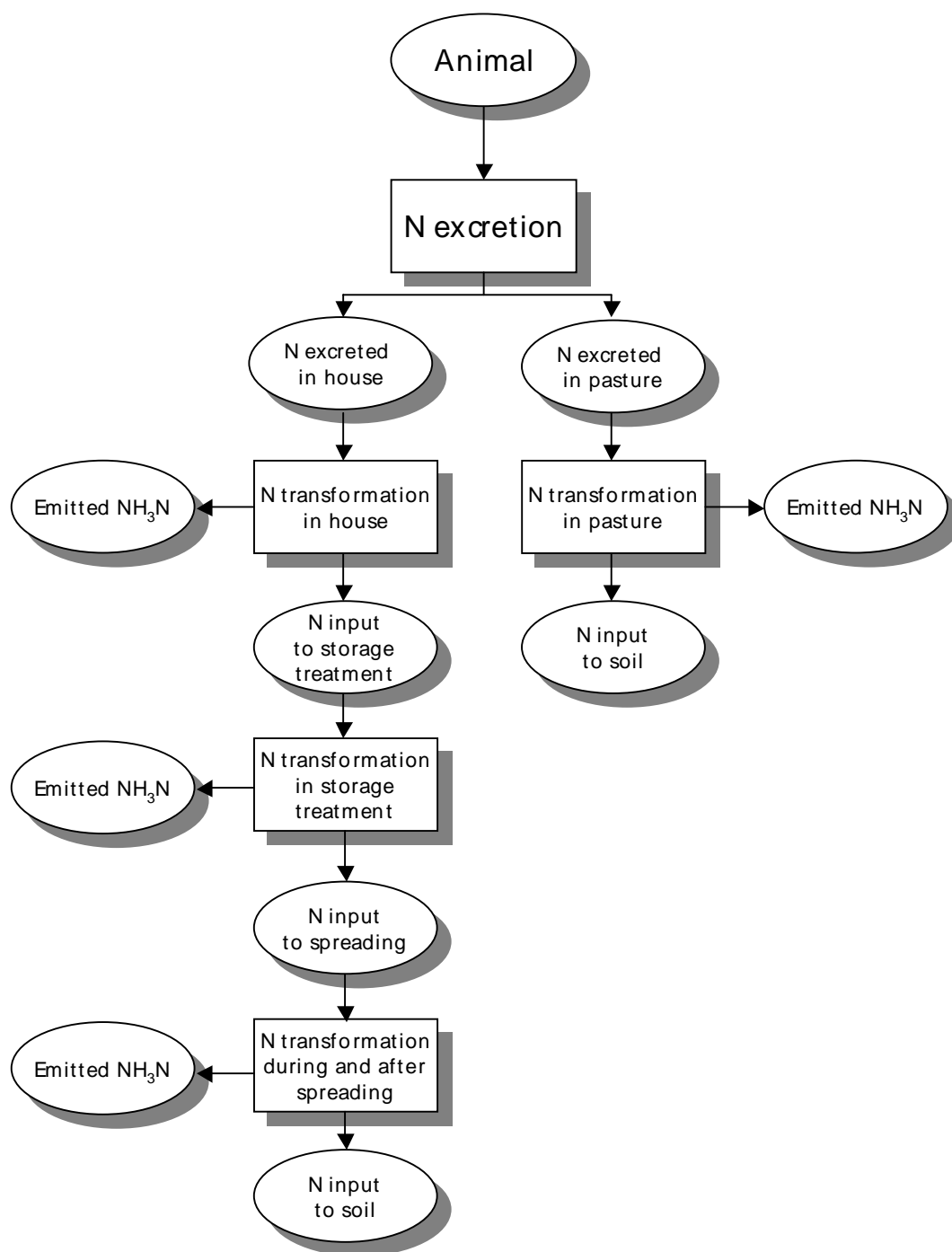
Figure 1. Likely Pathways for Animal Waste Ammonia Emissions to the Atmosphere

(Adapted from Hutchings et al., 2001)

Parameters influencing NH_3 emissions that relate to feed intake and excretion include:

- Feed intake, typically associated with the weight and role of the animal (e.g., breeding sow, farrow, finisher in the case of pigs) because, depending on their role, animals receive different feeds and have different weights and nitrogen metabolisms;
- Nitrogen content of the feed;
- Nitrogen content of spilled feed and bedding or litter;
- Division of nitrogen over urine and feces;
- Frequency of urination, and urine volume and nitrogen concentration;
- pH of urine and mixed manure and urine; and
- The conversion factor between the nitrogen in the animal feed and the nitrogen in the products (meat, eggs, milk) which determines the amount of nitrogen in the excreta.

Parameters influencing NH_3 emissions that relate to waste management include:

- The type of housing system, including ventilation, area per animal, and type of confinement structure, type of floor;
- The type of waste management including the storage and removal system within the house (e.g., pit storage, scrapers, frequency of animal waste removal);
- Additional nitrogen from spilled feed, or bedding, or litter;
- Waste treatment and disposal (lagoon, slurry tank, land application, composting, etc.);
- Meteorological conditions, including air temperature, air turbulence or wind speed, air humidity, and precipitation; and
- NH_3 concentration, pH, and surface water temperature in lagoon.

Parameters influencing NH_3 emissions from spray application fields include:

- Type of waste product that is applied (fresh animal manure, lagoon effluent);
- Amount applied per area;
- Concentration of NH_3 in the effluent;
- Frequency of application;
- Type of application (traveling gun, broadcast spreader, injection);
- Meteorological conditions, including air temperature, air turbulence or wind speed, air humidity, and precipitation; and

- Soil conditions (including permeability, porosity, pH, mineral content, moisture content) and soil vegetation cover. These factors determine the amount of nitrogen that is absorbed by the soil and crops that may be present.

Although the roles of the parameters determining NH_3 emission are well understood, there are insufficient scientific data to quantify the processes in detail and insufficient statistical data to quantify all input variables (Hutchings et al., 2001). As such, the model or emission factor(s) used to estimate emissions should reflect the quantity and quality of data available. Consequently, it is important to clearly define the purpose of the emission estimates because this will determine what type of model should be developed or used. For example, estimates that are intended to determine NH_3 emissions at the farm or regional scale can likely be based on more detailed activity information (e.g., average nitrogen intake per animal) and will benefit from refinement by including additional parameters from the lists above. Global and national emission estimates must probably continue to rely on emission factors that make use of broad activity data (e.g., number of animals). In this case, the level of detail is not likely to go beyond animal categories and some sub-categories and possibly a few broad categories of waste management systems. Accuracy in these types of estimates would be greatly enhanced if mortality, vacancy, and average weight are taken into account (Groot Koerkamp, 1998).

2   REVIEW OF EUROPEAN AND GENERAL NORTH AMERICAN LITERATURE

2.1   CHRONOLOGICAL SYNOPSIS OF AMMONIA EMISSIONS RESEARCH

In Europe, excessive NH_3 deposition has been investigated extensively during the last 15 to 20 years, for its potential role in soil acidification and eutrophication (Battye et al., 1994; IKC/RIVM, 1995). Early U.S. research focused on the role of NH_3 in atmospheric models, because NH_3 is the main alkaline constituent in the atmosphere. For example, in the 1980s, NH_3 emissions were included in national inventories developed under the National Acid Precipitation Assessment Program (NAPAP). Schlesinger and Hartley (1992) developed a global NH_3 emission inventory using European emission factors and were among the first in the U.S. to focus attention on possible large-scale effects of nitrogen deposition such as nitrogen saturation and acidification of soils, quoting European research.

An EPA report was published in 1994 that compiled and reviewed literature on sources of NH_3 and NH_3 emission factors (Battye et al., 1994). The report concluded that most research on NH_3 emissions as it relates to acid deposition was concentrated in The Netherlands, Great Britain, and Denmark. It was further determined that the majority of NH_3 emissions in the U.S. originate from livestock waste (about 80%). Other than the NAPAP emission factors, which were deemed unreliable, no recent and more reliable U.S. emission factors were found at that time. The Battye report recommended that European animal waste NH_3 emission factors that were developed by Asman (1992) in Battye et al. (1994) also be used in the U.S.

Asman defined 21 animal categories and sub-categories and three broad waste management categories; i.e., stable and storage; spreading; and grazing. The emission factors in Asman were based on tests that were conducted in The Netherlands in the late 1980s by various researchers and were developed by dividing the emission of a category by the number of animals in that category. The main limitations of the Asman emission factors are that animal weight and climate factors (expressed in seasonal and diurnal temperature variation) are not taken into account (Van der Hoek, 1994). In addition, there are different animal waste management practices in Europe and the U.S., the main difference being that lagoons are uncommon in Europe. Instead, waste is stored in concrete tanks as one may still find in the U.S. Midwest. Given the above, it can be concluded that these Asman emission factors are not likely to be well suited for estimating U.S. emissions. Yet, during the late 1990s, they continued to be used in the U.S., for lack of better data.

In the U.S., it was not until 1998/99 that substantial U.S. NH_3 emissions data from field tests started to become available. Field tests were conducted in the mid to late 1990s at swine operations in North Carolina

and, to a lesser extent, in Georgia. Most of these field tests pertained to emissions from lagoons because lagoons were thought to be the largest contributors to NH_3 emissions compared to houses or spray fields. Two studies were done that provide emissions data from swine houses, while a Georgia study pertains to spray fields. Chapter 3 summarizes the results of these field tests. Another North Carolina study used modeling to develop a rough overall NH_3 emission factor for a commercial swine farm in North Carolina (McCulloch et al., 1998). Additional work was done in the early 1980s in California, and later in Germany and in the United Kingdom (Asman, 2001). Many other studies focus on human or veterinary health and provide concentrations of NH_3 and other pollutants inside swine houses; however, these studies typically do not include air flow rates so no emissions could be developed.

In reviewing mainstream, international scientific literature published between 1994 and 1999 a few European papers were found that included new field tests data on NH_3 emissions. In addition, several summary papers were found that provide reviews or NH_3 emission estimates using existing emission factors (Bouwman and Van der Hoek, 1997; Sutton et al., 1995). These and other review-type papers and the citations therein indicate nonetheless that emission factor research was ongoing, especially in The Netherlands and to a lesser degree in the United Kingdom and Denmark. Most of this research appeared to be focused on measuring emissions from different animal houses. For example, a paper published in the Journal of Agricultural Engineering Research (Groot Koerkamp et al., 1998) summarizes comprehensive NH_3 emissions field tests that were done on livestock buildings in The Netherlands, the United Kingdom, Denmark, and Germany. None of the independent test results cited in Groot Koerkamp appear to have been published in leading journals. Instead these studies look like doctoral theses or comparable efforts, which are not published outside of the coordinating university or government agency and mostly appear to be produced in the language of the country of origin. Because a relatively large portion of the European literature discusses emission from houses, a separate section (2.2) is devoted to this topic.

The reason why little information was found is probably because the NH_3 issue in Western Europe has been evolving over a longer period of time and is, therefore, thought to have matured. For example, NH_3 has been regarded as a problem in The Netherlands for over 20 years. Methodologies to estimate emissions at the farm or regional scale, as well as at the (multi-) national scale have been established and are widely accepted. The Dutch have developed a comprehensive mass balance approach, which is detailed separately in Chapter 4. In Western Europe, comprehensive control mechanisms, such as manure injection, were put in place some time ago. Any field tests that have been conducted after 1994 (the year the Battye report was published)

serve to refine the existing emission factors and are probably not considered interesting or novel enough for publication.

Another reason why limited literature pertinent to the estimation of U.S. NH_3 emissions from animal waste was found is that there are distinct differences in U.S. and European waste management practices. First of all, the use of lagoons has always been uncommon in Western Europe, due to lack of available space and high groundwater tables (in the case of The Netherlands). Instead, manure is usually stored in concrete basins and land-applied as a slurry. This system is comparable to older U.S. systems in the North and Midwest. In the early 1990s in The Netherlands and probably in most of Western Europe, regulations were issued that made covering of animal manure storage facilities mandatory. In addition, land-spraying by traveling guns (sprinklers), as is done in the U.S., has been discouraged in Western Europe since the late 1980s in favor of manure injection or spraying techniques close to the ground followed by immediate tilling. Once these storage and application emission sources were reduced, research focused on developing a better understanding of the remaining source (i.e., emissions from houses), which is addressed in Section 2.2.

Two British and one Canadian paper were found that provide interpretations of existing NH_3 emissions information to assess potential impacts to the home countries of the respective authors. Sutton et al. (1995) published a paper in 1995 entitled "Assessment of the Magnitude of Ammonia Emissions in the United Kingdom." It provides a detailed comparison of NH_3 emission literature from the 1980s and early 1990s but includes no new field research data. The study concludes that there remains a large uncertainty in NH_3 emissions estimates. According to Sutton et al., total emissions in the United Kingdom were 450 (231-715) gigagrams[4] NH_3 per year (Gg/year), with cattle contributing 245 Gg/year. In a later paper by Pain et al. (1998), these estimates were apparently adjusted downward. Pain et al. estimated NH_3 emissions from cattle in the United Kingdom to be 99 Gg/year, 49 Gg/year of which being from land application, 27 Gg/year from housing, and the rest from storage. Ammonia losses from poultry and pig production, and fertilizer use were estimated at 30, 23, and 32 Gg/year, respectively.

A Canadian assessment of available NH_3 volatilization data was done by Paul (1997). For example, using a nitrogen mass balance approach on a swine house in Ontario, one Canadian research team estimated that 43% of the excreted nitrogen was lost as NH_3. This could be converted into an emission factor of 4.6 to 7.0 mg nitrogen per hr per kg live weight. Paul (1997) reported that NH_3 volatilization from pig houses was 37 to 40% of excreted N. Danish research reported in Paul found average NH_3 volatilization of 35, 20, and 40%

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[4]   A gigagram equals 1 kilotonne or 1 million kilograms.

from animal houses, manure storage facilities (concrete tanks), and land application, respectively. In summary, Paul concludes that between 40 and 95% of excreted nitrogen may be volatilized as NH_3 before the manure reaches the field.

Ammonia volatilization from land application of pig slurry in France (Moal et al., 1995) was estimated to be between 37 and 63% of ammoniacal nitrogen. Lorimor (1999) reports an even greater range for NH_3 losses from land application of pig waste; i.e., 11 to 78% ammoniacal nitrogen. The proportion of NH_3 varies from high Total Kjeldahl Nitrogen (TKN[5]) (larger than 80%) in lagoons, to around 65% in slurry pits, and down to around 15% in solid manure. According to Lorimor, there is typically no nitrate in manure. Most of the ammoniacal nitrogen spread on land will volatilize rapidly (e.g., in a few hours to a day); whereas, the organically bound nitrogen does not volatilize. The volatilization rate depends on the weather, with increased volatilization during warm, breezy conditions. Also, losses may be greater from crop residue than from tilled soil, because the positive NH_4^+ ions can cling to negative soil particles (Lorimor, 1999).

2.1.1   Dutch and Danish Information

Most work on NH_3 emissions from animal waste has been done in The Netherlands. This is not surprising because The Netherlands has a very high concentration of farm animals. In 1998, there were 15 million swine, 4.6 million cows, and 81 million poultry (Cowling et al., 1998). The Country has an area of 40,000 km^2 (15,400 mi^2), or a little less than one-third the size of North Carolina. Ninety-five percent of NH_3 emissions in the Country come from animal waste. Ammonia emissions peaked in 1987 at around 250 million kg NH_3 per year to level of 152 million kg NH_3 per year in 1995. The reduction in emissions is the result of control measures that include manure storage basin covers and low emission land application techniques, such as injection.

The potentially acidifying qualities of NH_3 were first recognized in The Netherlands in 1982. In 1984, a large coordinated research program was launched, reflecting a growing concern for the potential damage to forests. The first two phases of this research program resulted in an understanding of the causes and effects of the deposition of acidifying pollutants on forests and heathlands. Long-term emission targets were derived from critical loads for various types of ecosystems and from targets for nitrate leaching in natural areas (max. 25 mg/L). It was found that the protection of forests and heathlands required a deposition limit of 1,400 moles of H^+ per hectare and 1,000 moles of N per hectare. This would imply major reductions that

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[5]   Total Kjeldahl Nitrogen represents the total ammonia and organic nitrogen in a sample and is determined by digestion where organic nitrogen is converted to ammonia (Metcalf & Eddy, 1991, p. 85).

were to be reached in phases. The first phase was a reduction of 30% compared to 1980 levels. Ultimately, the goal is a 70% reduction in 2005, which translates into a national NH_3 emission of about 60 million kg NH_3 per year. It is now unlikely that this goal will be met (Hoogervorst, 1997).

As part of the aforementioned broad program, the Dutch Institute for Health and Environment (RIVM) developed a comprehensive methodology to estimate NH_3 emissions from animal manure, fertilizer usage, industrial processes, and households. This methodology was published in a 50-page document in 1994 (Van der Hoek, 1994). In 1998, the NH_3 emission methodology underwent a comprehensive review by researchers associated with the Ministry of Agriculture and Fisheries. The reason was that the level and trend of estimated emissions from agricultural sources were consistently significantly different from emissions calculated from ambient air data. The comprehensive review included the most recent literature and expert knowledge available in The Netherlands at that time. The results of this study are detailed in a 138-page document (Steenvoorden et al., 1999) and are summarized in Chapter 4, together with text from the methodology report (Van der Hoek, 1994).

Also, the Danish Government has opted for a mass balance approach. Danish NH_3 emissions were estimated by Hutchings et al. (2001) who applied a mass-balance-based model in a well-documented and comprehensive effort that is summarized here. The model that was used was developed to match the quality and quantity of available activity data in Denmark. Ammonia emissions are calculated separately for housing, manure storage, and during and after spreading (see Figure 1). Animals were divided into 31 categories according to species and housing type. The categories were chosen to match available national activity data from the Danish Agricultural Advisory Centre. Next, the total Danish livestock and poultry population was distributed amongst these categories, and the total nitrogen excreted annually by the animals in each category was calculated by multiplying the animal numbers with the annual nitrogen excretion per animal. The fate of this nitrogen was followed throughout the manure-handling chain, with NH_3 emission calculated as a percentage of the amount of nitrogen present in each link in the chain. The model also takes into account additional nitrogen from bedding and spilled feed. Emission factors are expressed as NH_3 nitrogen in percent of total remaining nitrogen. For example, emissions from spreading are expressed in percent of nitrogen present after storage.

In Denmark, emission factors are available for the main categories, but not for all categories. For example, according to Hutchings there are no well-documented studies on NH_3 emissions from poultry manure storage. As mentioned earlier, this Danish model distinguishes between emissions from manure spreading and emissions from after spreading. The emission factors for emissions after spreading take climate, crop cover, and seasons into account and are, again, based on a series of field tests. According to Hutchings et al. (2001), emissions from the act of spreading slurry-type manure with a broadcast-type spreader are minimal (i.e., 1%); whereas, emissions after spreading on a crop bearing field in summer can be as high as 30%. No further details were provided.

This Danish model shows that there are large differences in NH_3 emissions between different animal species and manure handling systems. Animal husbandry systems that make use of litter (e.g., hay) have higher volatilization rates than systems that rely on slurry. Hutchings points out that substantial uncertainties are associated with emissions from litter systems, with the allocation of animals with housing types, and manure handling systems. Other issues are associated with the fine-scale geographical allocation of emissions. NH_3 has a relatively high deposition rate, so local agricultural sources can have a large effect on neighboring ecosystems. Furthermore, Hutchings emphasizes that the model will be valid only as long as the chemical and physical characteristics of the manure don't change. The model cannot be used to examine the effect of changing these characteristics; e.g., changing feeding practices or the pH of the manure. The model is better suited for modeling the effects of alternate distributions of animal populations over the different categories.

2.2   EMISSIONS FROM HOUSES

Two comprehensive papers in the English language were found that include field test information. A paper by Groot Koerkamp (1998) is summarized below. It is illustrative of the research that has been conducted in Europe in the last 5 years, focusing on emissions from houses. The second paper, by Demmers et al. (1999), describes measurements quantifying the NH_3 emissions from an English broiler chicken house and a finishing pig unit. The NH_3 was converted to nitric oxide the concentration of which was analyzed using chemiluminescence. Other papers that contained novel field test results are Hendriks et al. (1998) and Aarnink (1997), and these also detail emissions from houses. Pertinent findings of these papers are included in Section 2.2.1, entitled: "Mechanisms Governing Potential Ammonia Emissions Reductions from Swine Houses."

Groot Koerkamp (1998) includes a discussion of NH_3 emissions field test results from various housing types for different livestock in The Netherlands, United Kingdom, Denmark, and Germany. Fourteen combinations of livestock and housing were defined that could be found in each of the four countries. Measurements were conducted at seven different sampling points in each house over a 24-hour period in summer and winter at each of the approximately 56 (14 x 4) locations. Data collected included concentration, air flux, and temperature, as well as animal counts and weights. The NH_3 analyzer used was a combination of a chemiluminescence nitrogen oxide analyzer and a thermal NH_3 converter. The seven sampling points were in a cross-section in the middle of the house, three at about 1.5 m height, three at 2.5 m height, and one close to the exhaust.

Ammonia emissions are expressed in milligrams (mg) per hour per animal, per 500 kg live weight, and per heat producing unit[6] (hpu) where 1 hpu = 1 kW. Tables 1 and 2 include emissions per animal and per 500 kg live weight, as well as the variance between summer and winter emissions and between replicates. The estimated NH_3 emission for each location was corrected for the mean outside temperature per country. The mean emission of NH_3 over 24 hr. was assumed to have a Poisson distribution. The effect of seasonal variations on the NH_3 emissions of the outdoor temperature was generally between -5 and +5% per °C.

Groot Koerkamp concludes that not all variations could be explained in terms of physical and chemical processes involved in the emission of NH_3. There are variations between countries for the same animal species and the same housing type; between replicates of a certain housing type in a country; diurnal variation; as well as yearly or seasonal variation beyond what may be expected from a relationship between outdoor temperature and emissions. The spatial variation of the seven sampling points inside a house was relatively small compared with the other sources of variation. The variations are in part due to the fact that the effect of the manure handling (daily, weekly, or monthly removal from the building) and the effect of growth of the animals during the production period on NH_3 emissions were not taken into account in the statistical analyses. (It is assumed that the relationship between size of the animal and NH_3 emissions is linear.) It is possible that a large part of the variations may be caused by differences in diet (expert judgment by authors).

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[6]   No other information was available regarding this parameter, and it was not used by other authors.

Table 1: Ammonia Emissions for Different Livestock and Housing Systems in England, The Netherlands, Denmark, and Germany (live weight based)

	Ammonia Emissions in mg NH_3/hour/500 kg live weight							
	England		The Netherlands		Denmark		Germany	
Animal type and housing system	Mean	c.v. %	Mean	c.v. %	Mean	c.v. %	Mean	c.v. %
Dairy cows, litter	260	42	890	24	491	19	467	30
Dairy cows, cubicles	1048	49	1,769	23	843	20	1,168	30
Beef cows, litter	478	44	-	-	-	-	431	26
Beef cows, slats[a]	-	-	853	23	900	17	371	30
Calves litter	315	39	-	-	1037	18	886	23
Calves, slats/group	-	-	1,148	23	-	-	1,797	23
Sows, litter	744	38	-	-	-	-	3,248	34
Sows, slats	1,049	38	1,282	24	1,701	17	1,212	24
Farrows, slats	1,047	38	786	27	1,562	18	649	25
Finishers, litter	1,429	39	-	-	3,751	20	-	-
Finishers, slats	2,592	39	2,076	23	2,568	18	2,398	21
Layers, litter	7,392	38	9,455	23	10,892	20	-	-
Layers, deep pit	9,316	38	1,624	26	2,160	34	602	28
Broilers, litter	8,294	41	4,179	24	2,208	33	7,499	24
Variance (cv %)								
between replicates	68		42		8		21	
between seasons	46		24		46		57	

From Groot Koerkamp (1998)

[a] slats in barn floor

The values presented in Groot Koerkamp et al. must be considered as the mean emission rates for mean conditions of manure handling and growing stage of the animals. The authors are of the opinion that the disadvantage of the short (24-hr) measuring period in each house was overcome by the number of repetitions of measurements in four replicates of each housing type under summer and winter conditions. A comparison with Dutch data showed that the measurement method for NH_3 emissions used in this research produces accurate mean emission rates per animal and housing type in the four countries.

Table 2: Ammonia Emissions for Different Livestock and Housing Systems in England, The Netherlands, Denmark, and Germany (animal based)

	Ammonia Emissions in mg NH₃/hour/animal							
	England		The Netherlands		Denmark		Germany	
Animal type and housing system	Mean	c.v. %	Mean	c.v. %	Mean	c.v. %	Mean	c.v. %
Dairy cows, litter	314	45	974	24	56	24	538	31
Dairy cows, cubicles	1245	52	2,001	24	987	25	1,320	31
Beef cows, litter	482	48	-	-	-	-	262	27
Beef cows, slats[a]	-	-	686	24	580	22	346	31
Calves litter	80	41	-	-	332	23	193	24
Calves, slats/group	-	-	522	24	-	-	323	24
Sows, litter	303	40	-	-	-	-	1,298	35
Sows, slats	503	40	535	24	730	23	325	25
Farrows, slats	26	40	27	28	46	23	22	26
Finishers, litter	108	42	-	-	394	26	-	
Finishers, slats	185	42	385	23	319	23	308	22
Layers, litter	31	41	36	24	38	26	-	
Layers, deep pit	39	40	6	27	8	40	2	29
Broilers, litter	20	44	11	24	9	44	19	25
Variance (cv %) Between replicates Between seasons	69 55		43 26		28 48		22 58	

From Groot Koerkamp (1998)

[a] slats in barn floor

Demmers et al. (1999) estimated an emission factor of 47 kg NH₃ per 500 kg live weight per year for swine, based on continuous occupation. Because the average weight of the animals was 25.7 kg, this can also be expressed as 270 mg NH₃/animal/hour. In contrast, Groot Koerkamp et al. found 185 mg/animal/hour for English finishing pigs on fully slatted floors (see Table 2), as well as 385, 319, and 308 mg/animal/hour for pigs in The Netherlands, Denmark, and Germany, respectively. The emission factor value reported by Demmers for broilers is 17 kg NH₃/500 kg/year; based on a 290-day occupation during the year this can be converted into 2,442 mg NH₃/500 kg/hour. This value is much lower than the Groot Koerkamp's 8,294 value for English broilers but does compare well to the Danish number (see Table 1).

2.2.1 Mechanisms Governing Potential Ammonia Emissions Reductions from Swine Houses

There are numerous control mechanisms and technologies that have been proven or have the potential to reduce NH₃ emissions from swine waste. Example technologies with a proven track record include more frequent and separate removal of urine and feces, storage basin covers, and swine waste injection into soil. A discussion on these technologies is not part of this report, but from the literature on emissions, it became

clear that there are five principal mechanisms that govern NH_3 emissions from houses. These mechanisms are: nutrition, temperature, surface area of the waste collection pit under the floor, slatted floor type, and diurnal variations. Except for nutrition, these parameters are discussed below in more detail.

Animal nutrition is an important mechanism for controlling NH_3 emissions. By selecting certain feed, nitrogen excretion can be influenced. The type of nitrogen (i.e., mineral nitrogen or organic nitrogen) can be influenced, as can the fractions of nitrogen excreted via urine and feces. A detailed analysis of animal nutrition is beyond the scope of this report. Some general text is included in Section 3.1. Also, ample specialized information can be found in handbooks and in the literature. North Carolina State University employs several swine nutrition extension specialists that may be consulted (website: http://www.cals.ncsu.edu/an_sci/extension/).

--Temperature

Steenvoorden et al. (1999) describe research that was done in 1995 to study the relationship between NH_3 emissions and ambient air temperature inside a mechanically ventilated standard dairy house. Data are presented in Table 3 and Figure 2, which serve to illustrate that there is a significant positive correlation between NH_3 emissions, as well as volatilization percentage (calculated by assuming constant nitrogen quantities in excretion) and temperature.[7] However, it is not recommended to extrapolate these or other data to situations with significantly higher temperatures, because of the non-linear, exponential relationship between vapor pressure and temperature, as well as the influence of other parameters (Asman, 2001).

Table 3: Correlation Between Ammonia Volatilization and Temperature Inside Dairy House

Month (1989)	Temperature (ºC)	(ºF)	Ammonia Emission (g N/animal/day)	Volatilization [a] (%)
January	11.8	53	25.6	7.4
February	12.4	54	28.4	8.1
March	14.4	58	29.1	8.3
April	14.1	57	30.1	8.3
May	18.4	65	39.9	11.3

[a] Constant N-excretion of 352 g N/animal

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[7] Other research mentioned in Steenvoorden et al. (1999) provides additional qualitative proof of the effect of temperature, by discussing the benefits of refrigeration of swine waste in a house as an effective control technology to reduce emissions.

The chart contains:
- Y-axis: Volatilization (g NH3/aap/day)
- X-axis: Temperature (deg C)
- Equation: $y = 12.434e^{0.0626x}$
- $R^2 = 0.9495$
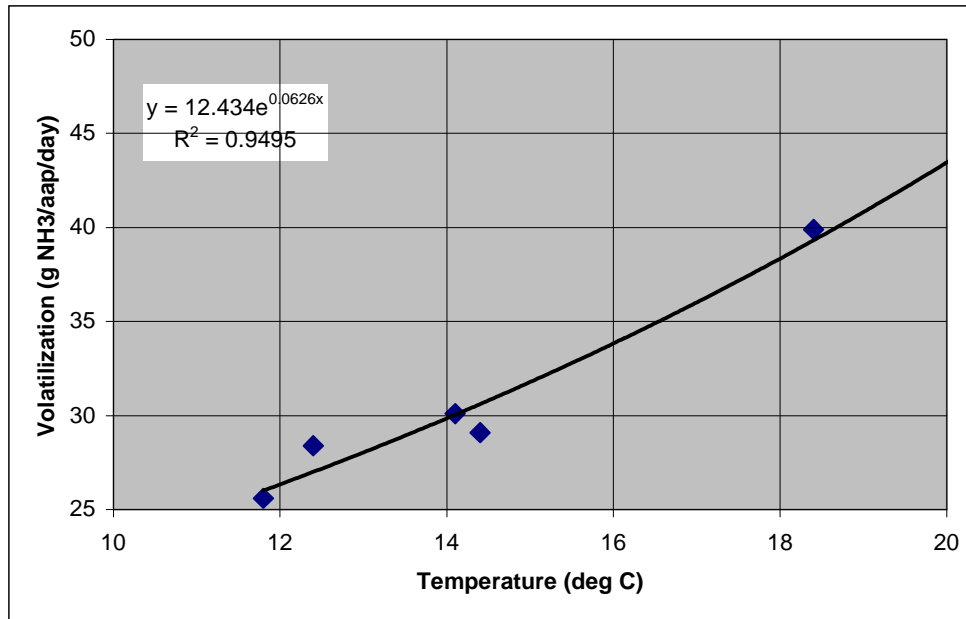
Figure 2: Relationship Between Ammonia Emissions and Ambient Temperature in a Dairy House

Another illustrative document is that by Hendriks et al. (1998), who studied the effect on NH_3 emissions of a biological additive to manure stored in a Belgian commercial swine house. The house was mechanically ventilated and partially slatted. Manure was stored in a pit 2 meters deep that was emptied twice a year. The effectiveness of the additive is not pertinent to this report, but there are interesting supporting data with standard deviations from thorough control tests, which are reflected in Table 4. During the periods indicated in Table 4, measurements were taken every 12 minutes. Emissions are lowest for the second fall/winter test. During this period the temperature above the manure was lowest. During periods of low temperature, the building is probably ventilated less, which will further contribute to a reduction in NH_3 volatilization. Also Aarnink (1997) noted seasonal variations in NH_3 emissions, with emissions generally higher in summer than in winter. An additional finding of Aarnink was that the solid floor (in a pen with a partially slatted floor) was fouled more during summer than in winter. An explanation for this behavior may be that the pigs prefer to lie on the cooler slats in summer, thereby fouling the other, solid side of the pen (Harris, 2001).

Table 4: Ammonia Emission Factors for a Belgian Swine House

Time period	Late summer (8/95 - 9/95)	Winter (12/94 - 3/95)	Fall to winter (9/96 - 2/97)	Fall to early winter (9/96 - 1/97)
NH_3 emission per animal (g/hr/animal)	0.5 (0.1)[a]	0.43 (0.18)	0.46 (0.20)	0.29 (0.14)
Average weight of animals (kg)	75	61	77	26
NH_3 emission per kg live weight (mg/hour/kg)	7.2 (1.9)	6.3 (2.2)	6.6 (3.5)	4.0 (1.6)
NH_3 emission per kg live weight (g/year/kg)	63	55	58	35
Temperature in pit (5 cm above manure level) (°C)	22.1 (1.5)	19.2 (2.0)	16.8 (3.1)	14.2 (1.2)
Relative humidity (%)	67 (6)	65 (5)	68 (10)	73 (6)

[a]   Standard deviation in parentheses

--Surface Area

Steenvoorden et al. (1999) also describe research to determine the correlation between the waste pit surface area that is fouled and NH_3 emissions per pig in a Dutch swine house (see Table 5). The findings concluded that NH_3 emissions show a strong positive correlation with waste pit area. A typical Dutch swine house has a scraper system and a partially slatted floor, which is different from the typical North Carolina pull-plug system where the entire house has a pit with water. Therefore, this research may not be directly relevant to the North Carolina situation.

Table 5: Relationship Between Ammonia Emissions and Waste Collection Area

Type of pig/ pit system	Pit area (m^2 per animal place)	Ammonia emission (kg NH_3 per animal per year)
Finishers		
• standard (50% slatted)	0.40	2.5
• separate manure gutters	0.29	1.8
• sloping floors	0.18	1.0
Gestating sows		
• standard individual confinement	1.1	4.2
• narrow manure gutter with metal slatted floor	0.4	2.4
Farrowing sows		
• standard fully slatted floor	4.1	8.3
• shallow manure pit with gutter	0.8	4.0

For both rearing and fattening pigs (farrows and finishers), Aarnink (1997) compared pens with a 50% slatted floor area with pens that have a 25% slatted floor area. For rearing pigs, the NH_3 emission from the 25% slatted pens was 20% lower than from the 50% slatted pens. For fattening pigs, the difference was 10%. Aarnink concludes that NH_3 emissions were positively related to the urine-fouled area and to the frequency of urination. Reducing the slatted floor and slurry pit surface area in houses for rearing as well as fattening pigs decreased NH_3 emissions from the slurry pit.

--Slatted Floor Type

To determine the effect of the slatted floor construction on NH_3 emissions, Aarnink (1997) did field tests on the excretion behavior of fattening pigs on five types of slatted floors. In the experiment there were two concrete slatted floors, a cast iron slatted floor, and two floors whose metal slats were triangular in cross section (tip of triangle in cross section pointing down). One of the metal slats was partially covered with studs to prevent the pigs from lying in the excreting area.[8] The studs were 5 cm high, 3.2 cm in diameter, and spaced 20 cm apart. The NH_3 volatilization from the metal slatted floors was significantly lower by 27%, compared to the volatilization from the standard concrete slatted floors with 10-cm wide slats and 2-cm wide gaps. The best result (least fouling) was achieved with the floor with studs, which had a 35% emissions reduction compared to the standard concrete floor (10-cm wide slats with 2-cm wide gaps). It was concluded that slatted floors from smoother material and with more open space than concrete slatted floors, such as floors with triangular section metal slates, significantly reduce NH_3 emissions from the slats.

--Diurnal Variations

Aarnink (1997) found that diurnal patterns in NH_3 emissions differed between houses with rearing pigs and houses with fattening pigs. Both houses had higher emissions during the day than during the night: +10% for rearing and +7% for fattening pigs. For rearing pigs, emissions peaked in the morning, but for fattening pigs they peaked in the afternoon. Aarnink suggests that this seems to be related to the behavior of the animals. Also, Harris (2001) noted a significant diurnal cycle in NH_3 emissions (see Section 3.1).

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[8] By nature, pigs are animals that prefer not to foul their resting area. When placed in a new pen, they first choose their lying area. The excretion area is generally located as far as possible from the lying area. However, the lying and excreting behavior of pigs is strongly influenced by the indoor climate. At higher ambient temperatures, the behavior of the pigs seems to be driven by finding cool spots to lie on, such as the slats (Aarnink, 1997).

2.3   REGIONAL AND GLOBAL METHODOLOGIES

Several efforts to estimate global NH_3 emissions have been conducted over the past 20 years. The most recent, comprehensive effort by Dutch, Danish, and English researchers in Global Biochemical Cycles that includes nine major NH_3 sources and was specifically developed for input into global atmospheric models (Bouwman et al., 1997). In this effort, emissions from domestic animal waste were estimated to be 21.6 teragrams N/year (Tg N/year); whereas, total global emissions from all sources were estimated at 54 Tg N/year. The overall uncertainty in the global estimate is stated to be 25%, while the uncertainty in regional emissions is much greater. The calculation of NH_3 emissions from domestic animal waste is based upon a mass balance method that uses average nitrogen excretion for different domestic animal categories and subsequent NH_3 losses during housing, storage, and land application, or grazing. Emission factors were based on the work of Van der Hoek and Couling (1996) and Van der Hoek (1998).

In another article in Atmospheric Environment by Bouwman and Van der Hoek (1997), the same methodology is used to develop scenarios for NH_3 emissions from developing countries. (See Table 6.) The nitrogen excretion data that are represented are very broad. In developing countries, lower feeding levels and a lower N content of the feed result in a lower volatilization fraction of the N in animal waste. It was assumed by the authors that this is counteracted by the higher temperatures in developing countries, in comparison to developed countries.

Table 6: Default Nitrogen Excretion and Ammonia Volatilization Rates for Dairy and Beef Cattle, Swine, and Poultry

	Developing Countries			Developed Countries		
Category	N excretion (kg N /head/yr)	NH_3 - N loss (%)	E. F. [a] (kg NH_3 /head/yr)	N excretion (kg N /head/yr)[c]	NH_3 - N loss (%)	E.F. [a] (kg NH_3 /head/yr)
Dairy Cattle						
Feedlot [b]	40	36		50	36	
Pasture	20	15		30	8	
Total	60	29	21	80	26	25
Beef Cattle						
Feedlot	10	36		15	36	
Pasture	30	15		30	8	
Total	40	20	10	45	17	9
Pigs	11	36	5	11	36	5
Poultry	0.5	36	0.22	0.5	36	0.22

[a] Emission factors were calculated by the authors as follows: (N excretion) × (N loss through volatilization) × 17/14.
[b] Based on open lot system, where cattle spend 40/60 = 66% of time in feedlot and 33% in pasture.
[c] N excretion data for developed countries are fairly low. This may be explained by the fact that "developed countries" also include countries that do not have highly developed animal management practices.

Another major effort to estimate European and country-specific NH_3 emissions was undertaken under auspices of the European Union as part of a comprehensive program that covers 28 European countries, 11 major source categories, and 8 air pollutants. The inventory is known as the EMEP/CORINAIR90 inventory and the emission factors and other data are detailed in the Atmospheric Emission Inventory Guidebook (AEIG, 1998), which is available on CD-ROM and on the Internet[9]. The NH_3 section can be found in the chapter entitled: "Agriculture and Forestry, Manure Management." As part of this effort, eight expert panels were established that provided input on various pollutants. The NH_3 emissions panel, that included 32 scientists from 17 European countries, defined default NH_3 emission factors that are applicable to the average European situation (see Figure 3).

Figure 3. European Union Ammonia Emission Factors

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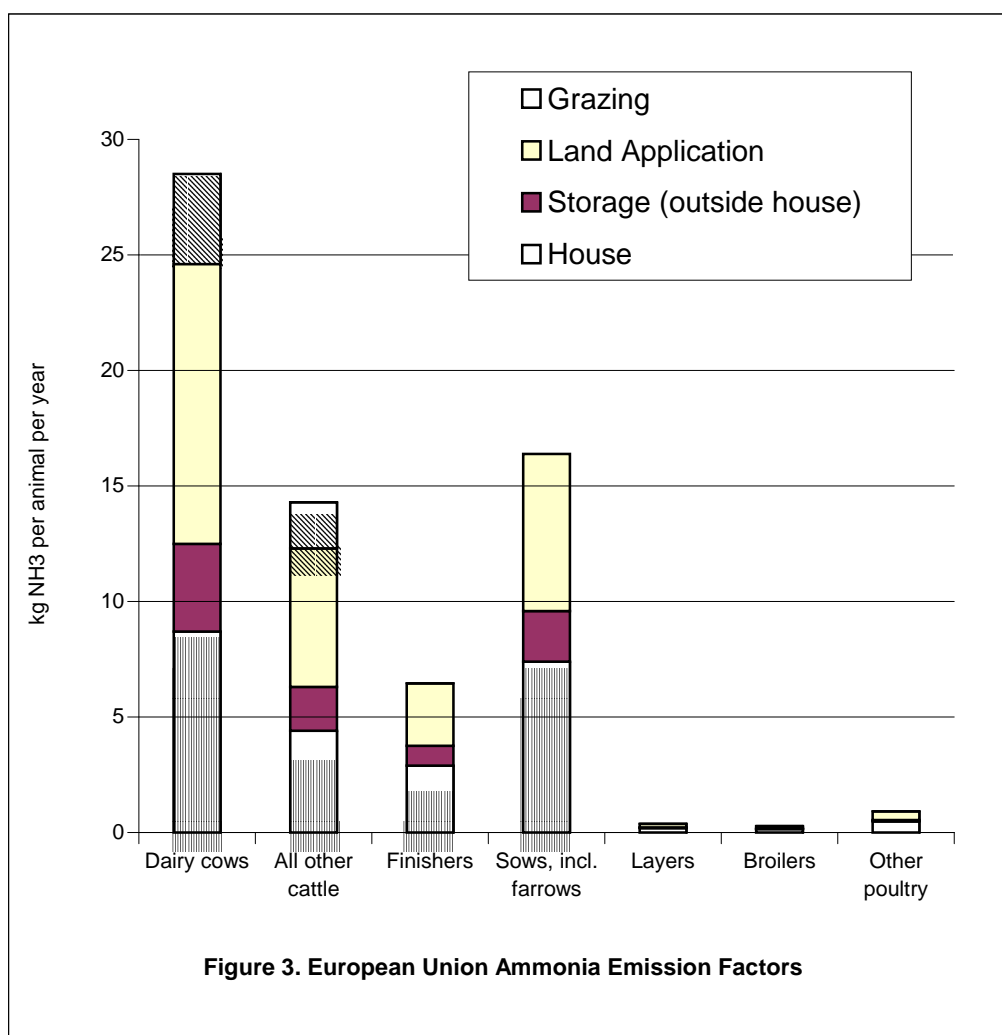
9 The website of the European Environment Agency is: http://www.eea.eu.int/. The website for the CORINAIR Atmospheric Emission Inventory Guidebook is: http://reports.eea.eu.int/EMEPCORINAIR/en/.

The European Union default emission factors are based on nitrogen excretions and volatilization percentages. The appendix to the Agriculture and Forestry, Manure Management chapter includes a detailed list of animal category-specific nitrogen concentrations and volatilization rates that were used to calculate the emission factors in Table 6. Countries that have more detailed data available can use this table to calculate more accurate emission factors. Unfortunately, no emission factors for lagoons are included, because lagoons are uncommon in Europe.

Asman's original emission factors are compared in Table 7 with those from Table 6 (see also Figure 3). The emission factors from Bouwman et al. (1997) are consistently somewhat lower than those of EMEP/CORINAIR. Only for beef cattle, is there a major difference among the three data sets, which may be due to differences in feed. The emission factors in Table 6 and Figure 3 do not differ markedly from the emission factors by Asman in Battye et al. (1994).

Table 7: Comparison in the Literature of Ammonia Emission Factors for Cattle, Swine, and Poultry [a,b,c,d]

	Asman in Battye et al., 1994				Bouwman & Van der Hoek, 1997	EMEP/ CORINAIR
	Stable and Storage	Land Application	Grazing	Total	Total	Total
Cattle (dairy)	7.4	12.2	3.4	23	25	28
Cattle (other)	7.4	12.2	3.4	23	9	14
Swine	2.5	2.8		5.4	5	6
Poultry (layers)	0.1	0.15		0.25	0.22	0.37
Poultry (broilers)	0.1	0.15		0.25	0.22	0.28

[a]   Emission factors are in kg NH_3 per animal per year.
[b]   These emission factors are for country estimates, not for individual farms.
[c]   The use of decimals does not indicate accuracy.
[d]   Sum of individual numbers may be different from totals due to rounding.

3   FIELD TESTS IN NORTH CAROLINA

Comprehensive field tests were conducted in the mid to late 1990s at a swine operation in Eastern North Carolina (Farm 10). The tests program at Farm 10 was coordinated by NCDENR and included research teams from or funded by NCDENR, USEPA's Air Pollution Prevention and Control Division (APPCD), North Carolina State University, the U.S. Department of Agriculture, and the University of North Carolina at Chapel Hill. Farm 10 is an integrated farrow-to-finish farm with nine finishing houses and four farrowing houses. The waste management system is "flush-type" with a pit under each side of the house, running the length of the house. Each pit (per half house) is flushed every week (assumed) for several hours with water from the lagoon. After flushing, no water remains in the pits. This type of waste removal system is uncommon, because most farms now have a pull-plug system[10] (see Appendix A). Effluent from the lagoon is sprayed on surrounding crop fields. The field tests focused on lagoons because in the mid-1990s, lagoons were thought to be the largest contributors to NH_3 emissions compared to houses or spray fields.

Three field tests (Aneja et al., 2000; Todd, 1999; and Harper and Sharpe, 1998) at Farm 10 pertained to emissions from lagoons, and the results are summarized in Section 3.2. In their field test report, Harper and Sharpe report on an additional, limited field test at another lagoon at a different farm (Farm 20). Another study at Farm 10 analyzed NH_3, NH_4^+ aerosol, and acid gas concentrations downwind of the farm and applied dispersion modeling to develop a rough overall NH_3 emission factor (McCulloch et al., 1998). Annular denuder systems were used to sample acid gases (hydrogen chloride, nitrous acid, nitric acid, sulfur dioxide) and NH_3, as well as fine aerosols (NH_4^+ chloride, NH_4^+ nitrate, and NH_4^+ sulfate). The study provides concentrations of all analytes and a coarse NH_3 emission factor for the whole farm of 5 to 10 kg per year per animal.

Unfortunately, no NH_3 emissions from spraying operations were measured for Farm 10; however, one Georgia field study was found that pertains to NH_3 emissions from spray fields (Sharpe and Harper, 1997). A micrometeorology method was used to determine NH_3 emissions from a sprayed oats field of 12 hectares in Georgia. To this field, 45 kg NH_3 as N per hectare was applied, of which 4.7 and 20.3 kg volatilized during application and post-application, respectively. This translates into a volatilization

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[10]   In this report, a mass balance approach is used to estimate total NH_3 emissions from a swine farm. The type of waste removal system in the house is not expected to be a significant parameter affecting total emissions. However, it is not recommended to take the isolated house emissions from Farm 10 and use them to estimate emissions from other houses. The same holds true for the lagoon emissions.

factor of 56%. Appendix B includes findings from a North Carolina study that focused on nitrous oxide emissions from spray fields.

Only one study was done at Farm 10 that provided limited sets of emissions data from swine houses (Harris and Thompson, 1998). The results of this study are included in Section 3.1. One of the conclusions from the comprehensive Farm 10 program was that additional emissions data were needed to better characterize emissions from swine houses. Especially, information was needed on diurnal and seasonal emissions, as well as emissions related to the size of the animals. As a followup to the Farm 10 program, the APPCD conducted the first phase of a comprehensive field study in 2000, at four separate, but nearly identical feeder-to-finish farms in southern North Carolina (Harris, 2001).

3.1   EMISSIONS FROM HOUSES

Harris and Thompson (1998) reported an NH_3 emission factor for several swine houses at the Farm 10 site in North Carolina. The houses at Farm 10 are equipped with five exhaust fans mounted at the lagoon-end of the houses, which turn on when the temperature exceeds 24 °C inside or by timer to allow for a minimum circulation of fresh air. Open-path Fourier transform infrared (OP-FTIR) spectroscopy was applied to collect data across a path behind nine houses, where air from the houses is exhausted by the fans. Because a fence ringed the houses, the exhaust ducts could not be sampled directly, and the infrared beam was aimed through the fan plume 1 meter from the duct exit. Emission factors were developed by monitoring the number of fans that were operating and estimating their flows from manufacturer's literature.

Using a finisher pig population of 6,000, Harris and Thompson (1998) report NH_3 emissions of 7.5 g/pig/ day for November 1997; 13.0 g/pig/day for January 1998; and 9.2 g/pig/day for May 1998, as well as an average emission factor of 9.9 g/pig/day. On an annual basis, these emissions are presented as 3.69 kg/pig/year with an individual seasonal range of 2.74 – 4.75 kg/pig/year. No data were collected for the farrowing houses. These values are based on certain assumptions used in the conversions from the measured ambient concentrations to the reported emission factors. These assumptions include:

- That emissions from the finishing houses were isolated from the farrowing houses by filtering the data based upon wind direction;
- A per-fan volumetric flow rate of 11,000 cubic feet per minute (cfm), which could not be adequately verified;

- The number of fans used in the calculations was based on a visual observation and was considered a constant for each sampling period; and

- Swine population was regarded as a constant for all sampling periods in the absence of hard data.

It should be noted that the values presented for Farm 10 are described as an "upper bound" since data were collected only during the daytime, due to both instrument limitations and restrictions from farm management. The research results do not take potential diurnal fluctuations into account, which could make the actual emission factor somewhat lower.[11]

As a followup to the Farm 10 program, the APPCD continued its research into emission factors for swine confinement buildings. In the first phase of a comprehensive field study in summer 2000, data were collected at four separate feeder-to-finish farms in southern North Carolina (Harris 2001). Each farm consisted of 10 tunnel-ventilated barns with a pull-plug waste removal system. Three barns at each farm were tested representing young, middle, and older age groups within the production cycle. Two chemiluminescent NH_3 analyzers sampled the exhaust from a rake mounted inside the exhaust cones. An anemometer determined air velocities at the optical path in the former study while the latter included calculations based upon fan duty cycle (sail switch determination), • P (differential pressure) measured across the fans, and the equipment manufacturer's factory calibration curves. Mass emission rate was determined from the product of total flow rate and concentrations measured.

Preliminary conclusions indicate that there is no statistically significant variation in the emission factor as a function of age or weight. The most likely explanation for this is that the recycled lagoon water used to flush the pit below the barn floor provides a baseline emission source that contributes a significant portion of the barn emissions. Also it is noted that there is a significant diurnal cycle. Based on these field tests a preliminary emission factor of 4.31 kg/pig/year is suggested for emissions in summer from pull-plug, feeder-to-finish operations.

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[11] Diurnal fluctuations may be less of an issue in poultry houses than in swine houses, because in poultry houses, the lights are on 24 hours per day.

3.2   EMISSIONS FROM LAGOONS

The lagoon at Farm 10 was sampled by several research groups over a period of a year using different techniques that include a flux chamber, a micrometereology method, and a FTIR technique. Summaries of the results of each group are given below, and detailed information on these different air sampling techniques is included in Appendix C.

Aneja et al. (2000) used a flux chamber method to measure NH_3 emissions from the lagoon surface. This method uses a plastic chamber of defined dimensions to isolate a portion of the source under investigation. During use, a zero-grade (zero background for the effluent being tested) compressed air source delivers a known flow rate through the chamber and carries the diluted effluent being tested to an appropriate analytical technique. (In this case, the NH_3 was converted to nitric oxide, which in turn was measured using chemiluminescence.) By measuring the effluent concentration in the diluted exit stream and having set the carrier gas flow rate, effluent mass per unit time is easily determined.   The researchers report that NH_3 flux displays a diurnal variation, which is highly correlated with lagoon surface water temperature. The average flux in the summer of 1997 was $4,017 \pm 987$ µg N/m^2 min; whereas, emission fluxes during the spring, fall, and winter were lower. Emission factors from this study are included in the summary table, Table 8.

A micrometeorology method was used by Harper and Sharpe (1998) to measure NH_3 emissions from the lagoon at Farm 10. The micrometeorology technique uses a vertical array of wind speed and temperature sensors operated with the air sampling occurring in parallel. During testing, this vertical array is floated to the middle of the lagoon. Ammonia concentrations were obtained by drawing unfiltered air through gas-washing bottles containing sulfuric acid at a known rate for 4 hours. The NH_4^+ concentrations were analyzed using colorimetry. The authors report that NH_3 emissions vary with time of day and season and were related to lagoon NH_3 concentration, pH, temperature, and wind turbulence. Emission factors from these studies are included in Table 8. In the field test report, the authors also discuss results from a field test at a lagoon at a second farm in North Carolina (Farm NC 20).

Todd (1999) used a tomographic open-path Fourier transform infrared spectroscopy (CAT-OP-FTIR) technology for measuring emissions from the lagoon at Farm 10. This technique requires two or more scanning OP-FTIRs and several retroreflectors. For the determination of emission rates, a tracer gas, non-reactive and without interference from ambient species or the effluent under study, was released from the

middle of the sampling area. Data were collected during daytime and evenings in November and May (see Table 8).

Table 8: Results from Ammonia Emissions Field Tests at Lagoons at Two North Carolina Swine Farms[a]

Field Test Method	Reference	Farm No.	Study Period	NH$_3$ per Lagoon	NH$_3$ per Animal	NH$_3$ per Standard Live Weight
				(kg/day)	(kg/ani./yr)	(kg/kg/yr)
Flux chamber	Aneja et al., 2000	**10**	Aug. 1997	156.2	5.64	0.0821
		10	Dec. 1997	32.8	1.19	0.0172
		10	Feb. 1998	11.9	0.43	0.0062
		10	May 1998	66.3	2.40	0.0349
		10	*Average*	*66.8*	*2.42*	*0.0351*
Micro-meteorology	Harper & Sharpe, 1998; Tables 1 and 3	**10**	Spring 1997 to winter 1998	28.1	0.75	0.0133
		10	Spring 1997	26.0	0.94	0.0137
		10	Summer 1997	50.5	1.82	0.0265
		10	Winter 1998	20.5	0.74	0.0107
		10	*Average*	*31.3*	*1.06*	*0.0161*
		20	Spring 1997 to winter 1998	14.7	1.25	0.0137
		20	Spring 1997	11.8	1.00	0.0112
		20	Summer 1997	13.8	1.17	0.0132
		20	Winter 1998	19.0	1.61	0.0182
		20	*Average*	*14.8*	*1.26*	*0.0141*
Flux chamber	Aneja et al. in Harper & Sharpe (1998)	**10**	Aug. 1997		1.80	
Tomographic OP-FTIR	Todd, 1999	**10**	Nov. 1997; daytime	70.5	2.55	0.0370
		10	Nov. 1997; evenings	89.2	3.22	0.0469
		10	May 1998; daytime	225.5	8.15	0.1185
		10	May 1998; evenings	274.8	9.93	0.1444
		10	*Average*	*165*	*5.96*	*0.0867*

[a]   Adapted from Aneja et al. (2000); Harper & Sharpe (1998); and Todd (1999). Refer to original papers for precision.

Farm:	Lagoon area:	Type of farm:	Waste removal:	Population:
No. 10	2.7 hectare	Farrow to finish	flush-type	7,480 finishers (135 lb); 1,212 sows, boars (400 lb); and 1,410 piglets (25 lb).
No. 20	2.4 hectare	Farrow to wean	pull-plug	2,352 piglets (25 lb), and 1,940 sows (400 lb).

4   MASS BALANCE METHODOLOGY

This chapter includes a comprehensive review of the mass balance methodology as it is applied in The Netherlands (Steenvoorden et al., 1999; Van der Hoek, 1994). Also, other countries in Europe, such as Denmark, have opted for a mass balance approach (Hutchings et al., 2001). NCDENR is conducting a preliminary evaluation of a simple nitrogen mass balance to document nutrient efficiency at the farm level. This mass balance would not distinguish between air and water emissions, but would merely estimate the portion of excreted nitrogen that cannot be accounted for (i.e., that is lost to the environment), while excluding molecular nitrogen (N_2). The mass balance uses a feed conversion rate for the nitrogen that is bound in the animal to calculate nitrogen output. From this number, the nitrogen that is emitted as N_2 gas and that is taken up by crops is subtracted. The methodologies for both the feed conversion rate and the N_2 gas conversion factor are undergoing an investigation to assess if more detail is needed to define these parameters adequately.

4.1   DUTCH METHODOLOGY

The Dutch NH_3 emission methodology follows a mass balance approach based on the average yearly nitrogen excretion per animal type and the different emission or volatilization factors from specific emission sources; i.e., house, storage/treatment, and land application. The nitrogen excretion is the difference between the nitrogen that is ingested by the animal and the nitrogen that is ultimately bound in the agricultural product (meat, eggs, dairy). For all animal categories, it was determined that possible nitrogen losses from hair, skin, sweat, and exhalation are negligible.

Because NH_3 emissions from the different sources will depend on the amount of nitrogen in the animal waste, emission factors are expressed in a percentage of the amount of nitrogen that is in the animal waste at that time at that source.[12] This approach takes the entire waste management pathway into account. For example, an emission reduction in the house may result in higher emissions from storage or land application,

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[12] This is not entirely correct, as was pointed out in Steenvoorden et al. (1999). From measurements and by modeling, it was shown that the relationship between nitrogen excretion and NH_3 emission is not linear. When the nitrogen concentration in the urine increases, the volatilization of this nitrogen will increase too. For example, for pigs, 65 - 70% of nitrogen may be excreted via urine, and 30 - 35% via feces. The feces nitrogen is mainly protein, either from the animal itself or undigested feed protein. In general it is assumed that this protein is digested slowly and NH_3 emissions (e.g., from houses) are limited. Therefore, it can be concluded that for pigs NH_3 emissions are more strongly correlated with the nitrogen in urine than with total nitrogen. (Steenvoorden et al., 1999) (This is not the case with chickens and other birds because all nitrogen is excreted via the kidneys.)

or a reduction of the amount of nitrogen in manure and/or urine would not necessarily lead to a proportional reduction in NH_3 emissions, because of the waste management method.

Emission factors were originally expressed in kg NH_3 per animal space per year. For example a farm with a capacity of 5,000 swine would have 5,000 animal spaces. This led to estimates that were too high because varying animal weights, mortality, and vacancy were not taken into account. As a result, a new parameter was introduced: average animal present (AAP). In the revised version of the Dutch methodology, NH_3 emission factors are expressed in percent volatilization of average nitrogen excretion (kg N) per AAP for each specific source.

For example, the methodology for calculating NH_3 emissions (E) from a specific type of house is:

$$E \quad = \quad n \times N_{ex} \times EF \ \ (\text{kg } NH_3 \text{ as N/yr}),$$

where:

$\quad$ n $\quad = \quad$ number of AAP,

$\quad N_{ex} \quad = \quad$ excretion per animal (kg N/yr),

$\quad$ EF $\quad = \quad$ percent volatilization of average nitrogen excretion (kg N) per AAP for that specific type of house.

When used with local animal population data, the Dutch model allows for estimation of NH_3 emissions, as well as estimation of animal waste generation. This is an advantage because, on a local or farm scale, information on total animal waste generation could be used to determine the number of animals in relation to factors, such as the required animal waste storage volume or application area. Larger scale NH_3 or animal waste data can be obtained by simple summation. An important assumption is that the number of animals for a certain geographical area during the count is equal to the average number of animals; i.e., the vacancy at the time of the count is equal to the average annual vacancy. Errors of this nature can be avoided by using AAP. Compared to "live weight," AAP also offers the advantage of better visualization to the user; 2,000,000 chickens present may be more meaningful to the general reader than 5,000,000 lb of live chicken weight.

4.1.1   Nitrogen Content

Steenvoorden et al., (1999) determined that a change in mineral content in the feed has a larger impact on mineral quantities in manure than a comparable increase or decrease in the number of animals. Mineral contents in feed can vary significantly, especially for cows, because cows move from stable to pasture and vice versa, depending on the season and the weather and will thus vary their diet from grass to hay and/or supplemental fodder. As far as the nitrogen content of the feed for each type of pig and chicken category, it was initially assumed that the nitrogen content does not vary significantly across The Netherlands. Later it was found that this assumption was incorrect (Steenvoorden et al., 1999). For example, if feed statistics from large operations ("integrators") are used across the board, nitrogen efficiency may be over-estimated. This is because large operations are more likely to make optimum use of feed and the nitrogen contained therein, whereas smaller operations may be less efficient. Also, certain farms may use feed of different quality that could influence the nitrogen content of the animal manure.

An important step to ensure the quality and increase acceptance of the Dutch model, was to reach consensus among scientists and stakeholders on manure and urine excretion data. Four factors are important in excretion calculations: the amount of ingested feed, the mineral concentrations in the feed, the amount of animal products (eggs, milk, and meat), and the mineral concentrations in these products (Van der Hoek, 1994). For NH_3 emissions work, only nitrogen compound concentrations are of concern.[13]

Animals within a certain category may not receive the same feed and produce manure of the same composition. For example, manure from boars or gestating sows will have different characteristics than that of finisher pigs. In the Dutch system, excretions from these smaller swine categories are normalized to finisher pigs, whereas feed intake and excretions of farrows are included with the lactating sows.

In the Dutch model, it is further assumed that the composition of feces and urine do not differ between day and night. The accuracy of this assumption would depend on differences in temperature, humidity, and wind from day to night, as well as feeding patterns. Pigs and poultry are typically in life-long confinement, but for cows on feedlots, it is necessary to determine what part of manure and urine is excreted in the stable and in pasture. Ammonia emissions per cow are lower for free-range cows compared to confined cows, because the urine can be absorbed by the soil, and there is less contact between urine and the manure which contains the urease.

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[13] Mass balances may also be applied for other minerals, such as phosphorus, potassium, and copper. Tracking these elements may prove to be beneficial for soil quality monitoring or other purposes.

Below, two examples are included to calculate nitrogen excretion for a finisher pig and a lactating sow (1992 data). As noted, similar calculations can also be done for phosphorus, copper, and other minerals.

Example nitrogen excretion calculation for finisher pig:

Feed intake	2.05 kg/day	or	748 kg/yr
Growth	0.717 kg/day	or	262 kg/yr (this is the meat production)
N in feed	26.3 g N/kg		
N intake	0.0263×748	$=$	19.7 kg N/yr
N in meat	23.0 g N/kg		
N fixation in meat	0.023×262	$=$	6.0 kg N/yr (30% of N intake)
N excretion	19.7 - 6.0	$=$	13.7 kg N /yr (70% of N intake)

Example nitrogen excretion calculation for lactating sow:

Farrows per lactating sow	20.5 farrows/yr
Feed intake farrow	30 kg/farrow
Feed intake all farrows	$20.5 \times 30 = 615$ kg/yr
Feed intake sow	1,097 kg/yr
Meat production farrows	551 kg/yr (weight of meat of all farrows per year)
N in feed for farrows	28.4 g N/kg
N in feed for sow	25.6 g N/kg
N intake from feed	$(0.0284 \times 615) + (0.0256 \times 1,097) = 45.6$ kg N/yr
N in meat	23.0 g N/kg
N fixation in meat	0.023×551 $= 12.7$ kg N/yr (28% of intake)
N excretion	$45.6 – 12.7$ $= 32.9$ kg N /yr (72% of intake)

4.1.2   Emissions From Houses

Some Dutch NH_3 emission factors for houses are presented in Table 9. In the last few years, nitrogen excretion numbers have been modified and better emission factors based on field measurements have been developed for many different types of houses. The new estimates are included in Table 9 in parentheses and illustrate the general downward trend in emission factors as a result of improved building and waste management designs. Significant efforts in control technology development have focused on the design of

new low-emission houses for all livestock categories. For example, Groot Koerkamp (1998) reported on field testing of 14 different housing types that were tested in The Netherlands, United Kingdom, Denmark, and Germany, which was summarized in Tables 1 and 2.

Table 9: Dutch Volatilization Percentages, Nitrogen Excretion and Ammonia Emission Factors for Cows, Swine, and Poultry in Different Housing Types. (Source: Steenvoorden et al., 1999)

Animal type	Housing type	Volatilization (%)	N-excretion kg N/AAP/yr	Emission Factor kg NH_3/AAP/yr
Dairy			61.3 [a]	
	Standard	14.6 *(9.4)* [b,c]		10.9 *(5.1 - 9.1)* [a,c]
	Confined housing	7.1 [b]		5.3 *(2.0 - 4.5)* [a]
	Low emission	9.5		7.1 *(5.3 - 10.0)* [a]
Cattle				
	Standard	12.6		
	Low emission	8.2		
Finishing pigs			14.5	
	Standard, fully slatted	19.3		3.1
	Standard, partially slatted	16.1 *(15.8 - 18.3)*		2.5 *(2.2 - 3.4)*
	Low emission (optimal house and feed)	13.0 *(11.9 - 14.2)*		2.1 *(1.0 - 3.0)*
Breeding pigs			29.9	
	Standard	19.5		7.1 *(5.6 - 10.8)*
	Low emission	13.0		4.7 *(1.6 - 7.5)*
Chickens (layers)			0.70	
	Standard (daily scraper)	8.7		0.07 *(0.08)*
	Standard (deep-pit)	40.5		0.34
	Floor with litter	18.7 *(8.3 - 20.6)*		0.16 *(0.05 - 0.13)*
	Low emission	3.7 *(3.5 - 5.1)*		0.03 *(0.01 - 0.03)*
Chickens (broilers)			0.59	
	Standard	10.6 *(9.9 - 11.2)*		0.08
	Low emission	2.6 *(6.7 - 7.4)*		0.02 *(0.005 - 0.014)*

[a]   Based on 190-day period;
[b]   Based on 365-day period; and
[c]   Data from additional, new research are included in *parentheses*.

The NH_3 emission factors in Table 9 are not the official Dutch emission factors. A comprehensive list of official emission factors was published by the Minister of Agriculture and the Minister responsible for the Environment in the Staatscourant[14] of December 16, 1998. This publication includes six pages of emission factors for all livestock categories. For pigs, there are official emission factors for 83 different subcategories for different types of pigs being held in different types of houses: traditional houses and various Green

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[14] Compare to Federal Register.

Label houses.[15] For example, in The Netherlands, finisher pigs are usually kept in pens on partially slatted floors. Gestating sows are kept in individual boxes with also a partially slatted floor. Lactating sows (mothers with piglets) are confined on totally slatted floors. Steenvoorden et al. (1999) comment that emissions per animal can vary significantly even for animals in the same type of house, although no ranges were provided.

The standard livestock management method in The Netherlands for layers (egg laying chickens) is rows of cages with the manure being removed by conveyors. Broilers are typically kept on floors with litter. There are significant differences between recommended NH_3 emission factors and those from more recent research as is indicated in Table 9. Dairy cattle are out in pasture during the warmer half of the year. In winter, the animals are kept in stables and can move about freely. The central walkway is slatted, whereas the bedding area is not. Recent volatilization and NH_3 estimates indicate that the currently used numbers are too high, as was indicated earlier in Table 9 (Steenvoorden et al., 1999).

4.1.3   Emissions From Swine Waste Storage

In The Netherlands, swine waste is typically stored in concrete storage basins. There are various collection systems for scraping and/or flushing of waste from the house. Lagoons are not used for several reasons, including limited space and high groundwater tables. As a result, the water that is used for flushing does not contain nitrogen. The storage basins are comparable to those found in older systems in the Midwestern U.S. Waste in these types of storage basins is significantly more concentrated than in lagoons. It is not recommended to apply Dutch NH_3 emission factors for manure storage to lagoons in North Carolina or elsewhere in the U.S.

In The Netherlands, in 1989 and 1990, "standard" emission factors were defined for NH_3 emissions from waste storage for all animal categories (see Table 10). These emission factors take into account calculated and actual storage periods (residence times) and account for storage covers, which were being introduced at that time. The control efficiency for "complete" coverage of a standard Dutch storage tank was determined to be 80%, implying there is still a significant amount of leakage.

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[15] A concept that has recently been introduced in The Netherlands is the "Green Label," which is a kind of verification approach of best management practices incorporated by farmers, animal house builders, and other stakeholders. This program has helped to encourage the design and incorporation of new low-emission housing systems. The Green Label concept is also being applied to other industries.

Table 10: Standard Emission Factors for Open Storage Basins in The Netherlands

	Emission Factor [a] (g NH$_3$ - N/yr/AAP)
Dairy cattle (while housed during winter)	3,460
Beef cattle (while housed during winter)	850
Finishing pigs	1,000
Farrowing sows	3,250
Broilers	12
Layers (cages)	100
Layers (cages, mechanical drying)	41
Layers (deep pit)	20
Layers (litter)	20

[a] These emission factors take into account standard and actual storage periods (residence times) and are corrected for duration and multiplied with a control effectiveness coefficient; e.g., for fitted for storage covers.

The emission factor to estimate NH$_3$ emissions from storage tanks for a certain region is:

$$EF = SEO \times AST/SST \times [CS \times (100 - CE) + (100 - CS)] \quad \text{g NH}_3\text{-N per AAP per year,}$$

where:

SEO	=	standard emission open storage, g NH$_3$-N/yr/AAP
AST	=	actual storage time, months
SST	=	standard storage time (6 months)
CS	=	percent of covered storage basins
CE	=	control efficiency for covered storage (80%)

Example calculation emissions from swine waste storage:

Actual storage time for finisher pig manure is 3 months; the standard emission factor is 1,000 g NH$_3$-N /year/AAP (from Table 10); 75% of farms in the specified geographical region have storage basins that are covered. Hence, EF = 1,000 × 3/6 × [75% × (100 - 80)% + (100 - 75)%] = 200 g NH$_3$-N per AAP per year.

Volatilization factors for animal waste storage:

Steenvoorden et al. (1999) includes another approach to estimating emissions from storage basins using volatilization percentages of nitrogen in stored manure. Volatilization percentages are provided for covered storage facilities for dairy cattle, finisher pigs, farrowing sows, and layers and are 0.96, 1.66, 2.36, and 2.57%, respectively. According to Steenvoorden there is sparse literature on NH$_3$ emissions from manure storage facilities. Williams and Nigro (1997) in Steenvoorden et al. (1999) reported an NH$_3$ emission rate of 2.3 grams per square meter (g/m^2) per day for an uncovered beef cattle manure storage site at 4 °C, which

climbed to 8.8 g/m^2 per day at 25 °C. Van der Meer (1991) in Steenvoorden et al. (1999) provided one additional generic number for beef cattle: 5 kg NH$_3$ N/year/animal. Sommer et al. (1993) conducted large-scale pilot tests and found rates of 3 to 5 g NH$_3$ – N/m^2/day for uncovered cattle and pig waste stored in open tanks, which were stirred once a week to prevent a crust from forming.

4.1.4   Emissions from Waste Application

In the Dutch system, the NH$_3$ emissions from animal waste application are expressed as a percentage of volatilized nitrogen in the animal waste. Nitrogen in animal waste can be classified as either organic nitrogen or mineral nitrogen, which can easily volatilize as NH$_3$. Mineral nitrogen is between 45 and 50% of total nitrogen in swine waste. Emissions field tests for different types of soil, with and without grass vegetation, were conducted in the late 1980s using micrometeorology and a mass balance approach. In the early 1990s, regulations were introduced that limit land application by spraying or spreading without further management of the soil. Most manure is now injected or applied to soil that is tilled within a short time frame; e.g., within 24 hours.

Initially, NH$_3$ emissions from land application[16] with rotating disc spreader were determined to be between 35 and 50% of mineral nitrogen (Van der Hoek, 1994). During more recent field measurements, higher volatilization percentages were found; e.g., 68% with very broad ranges (Steenvoorden et al., 1999). This value is somewhat higher than those reported in Paul (1997)[17]. One reason for the discrepancy may be that, due to currently existing regulations, more manure is spread during the spring and summer months, when the air temperatures are higher. Steenvoorden et al. (1999) reiterates that NH$_3$ volatilization during land application is strongly dependent on the weather at the time of application. Table 11 includes estimated and measured volatilization percentages for mineral (NH$_3$) nitrogen during land application of manure.

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[16] When the manure is not injected, land application (spreading) in The Netherlands is conducted by a spray system (rotating disc) that is placed behind a tanker trailer pulled by a tractor. Manure from the storage tank is spread fairly close to the ground. There are two other differences with the North Carolina situation: (1) in North Carolina, lagoon effluent is spread, as opposed to the more concentrated, fresher manure; and (2) in North Carolina, the effluent is sprayed higher into the air to maximize volatilization.

[17] Paul (1997) reported that NH$_3$ volatilization from pig houses was 37 to 40% of excreted N.

Table 11: Calculated and Measured Ammonia Volatilization Percentages During Land Application of Manure[a] in Western Europe (Source: Steenvoorden et al., 1999)

Technique	Calculated Volatilization (%)	Measured Volatilization [b,c] (%)	Volatilization Ranges (%)	Number of Measurements
Grass land				
• spreading	50	68	27 - 98	47
• injection	5	1	0 - 3	6
Bare soil				
• spreading	50	68	20 - 100	29
• injection	N/a	9	0 - 40	9
• spreading with immediate tilling	15	20	1 - 49	28
• spreading with tilling after 1 day	36	N/a	N/a	N/a

[a]   Not the same as lagoon effluent.
[b]   Percent of total ammoniacal nitrogen.
[c]   Measurements taken in spring, summer, and early fall.

4.1.5   Limitations to Mass Balance Approach

There are general limitations to the nitrogen balance approach. Inaccuracies in sampling of manure or litter, and inaccuracies in determining their nitrogen content can lead to inaccuracies in estimates of NH_3 losses. Although the mass balance method is universally accepted for most animal types, it may be best suited for poultry, because poultry feeding habits are fairly consistent (Van der Hoek, 1999). Improvements in the accuracy of nitrogen excretion calculations will influence the complete chain of NH_3 emission calculations. If the amount of nitrogen that is excreted is increased or reduced, more or less nitrogen is available for volatilization from the entire array of sources (house, storage, lagoon, spray field).

The NH_3 emissions from houses are calculated using the total nitrogen quantity in animal manure. Ammonia, however, is created from the breakdown of urea for mammals (or uric acids for poultry) and any mineral nitrogen (NH_4^+ compounds) in the feces. Only during storage or long-term presence of waste in the barn would organically bound nitrogen also be broken down to NH_3. In general, measures to decrease the urine nitrogen content, as opposed to the nitrogen content in feces, will lower NH_3 emissions, because of the slower reactions in the manure. Nevertheless, environmental parameters (see Section 1.2) may nullify any NH_3 control measures resulting from the feed, and it is important to review the complete interdependent nitrogen chain from feed to land application, including sludge handling (Steenvoorden et al., 1999, p. 33).

Another limitation of the Dutch mass balance method described above is that, in its current form, it is not equipped to address the loop that is induced by the use of NH_3-laden lagoon water to flush and fill the pit

under houses, as occurs in North Carolina in pull-plug houses. Lagoon water remains in the pit for up to a week, depending on the flushing cycle. As was indicated in Section 3.1, Harris (2001) found that the recycled lagoon water used to flush the pit below the barn floor provides a baseline emission source that contributes a significant portion of the barn emissions. In a mass balance approach for a pull-plug house, this additional source of nitrogen will have to be addressed. The situation at Farm 10 may be described as a hybrid system from a mass balance perspective, because the flushing water does not stay in the pit, but is directly returned to the lagoon by force of gravity. See Figure 4. It may be that in a flush-type house additional emissions from the recycled water are small compared to emissions from the fresh pig waste. In that case the Dutch mass balance approach may be appropriate for this type of farm. The method may also be useful as an emission estimation tool in the discussion of closing lagoons and alternative waste treatment methods.

Another omission in the mass balance approach is the potential NH_3 emissions from sludge treatment or spreading. In the Dutch system, the manure is spread entirely; whereas, in the North Carolina system only the lagoon effluent is spread, while the sludge continues to collect in the lagoon. This sludge will have to be treated or land-applied when the lagoon reaches its sludge holding capacity.
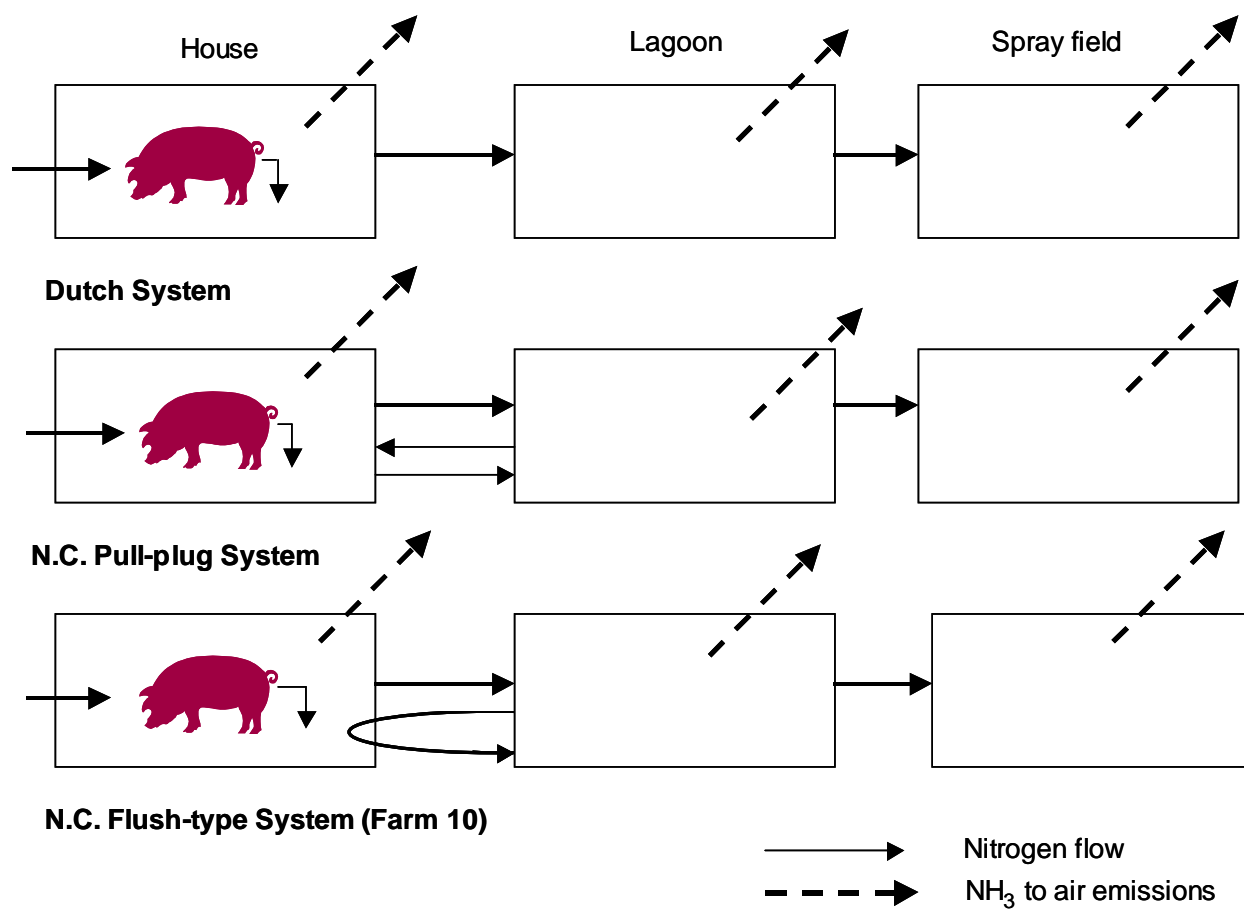
House   Lagoon   Spray field

Dutch System

N.C. Pull-plug System

N.C. Flush-type System (Farm 10)

Nitrogen flow

NH_3 to air emissions

Figure 4. Nitrogen Flows in Dutch and North Carolina Systems

5   COMPARISON AND DISCUSSION

To date, the most complete U.S. data set of NH$_3$ emissions based on field measurements from a full-scale swine farm is that of the North Carolina farm, "Farm 10" (see Section 3). In this section, the Farm 10 emission rates are compared with estimates based on the mass balance method as discussed in Section 4.0. Because, finishing pigs are the most significant sub-source category, and emissions from finishing pig houses only were collected at Farm 10, the finishing pig population was used as a base for the comparisons. Because no field tests were conducted at the farm's spray fields, an attempt is made to estimate these emissions based on volatilization percentages from the literature. Emission factors based on Farm 10 field test results are further compared to emission factors from the literature.

The total swine population at Farm 10 at the time of the field tests consisted of 7,480 finishers, 1,212 sows and boars, and 1,410 piglets; average weights are 135 lb (61.4 kg), 400 lb (181.8 kg), and 25 lb (11.4 kg), respectively. The total Farm 10 live weight was 1,529,850 lb (695,386 kg), and the average animal weight was 151 lb (69 kg).

5.1   MASS BALANCE APPLICATION

5.1.1   Simple Total Mass Balance

A simple estimate of NH$_3$ emissions can be based on average manure production values, manure nitrogen contents, and number of pigs. Barker (1998) provides values for average manure production by finishing pigs and the ammoniated nitrogen content thereof. Using a mean value of 11.1 lb of fresh manure per 135 lb finishing pig per day, 12.2 lb TKN per ton of manure, ammoniated nitrogen of 62% of TKN, and a farm population of 7,480 finishing pigs, we can arrive at an ammoniated nitrogen value of 143 kg per day for this finishing operation (see Table 12).

Much of this report, including the current section, was written from the point of view of a feeder-to-finish operation, which represents the majority of farms in North Carolina. However, the numbers calculated here and used to estimate emissions from spray fields, are specific to Farm 10, a farrow-to-finish operation. As such, we present the calculations for both the total farm and the feeder-to-finish part of the operation. Table 12 presents these additional estimates. Sows and boars are treated on the basis of weight ratio to 135 lb finishers (a factor of 3), and piglets are ignored due to their small population and small weight ratio relative to finishers. Finally, there is some disagreement as to the use of the 62% factor. This was discussed with Dr. Barker of North Carolina State University (NCSU). Dr. Barker said

that the "NH$_3$-N as a fraction of TKN" factor represents inorganic NH$_3$. He also agreed that it does include urea present in the urine. As such, it is an appropriate factor to apply when examining the houses in Section 5.2 (Barker, 2001). At the same time, it is unclear that any other factor is more defensible. After a literature search and discussions with NCSU, we have not found any other factor with a citable reference. Therefore, the best we can do at this time is to regard the 62% factor as a lower limit and 100% as an upper limit.

Table 12. Total Nitrogen Content of Waste Production at Farm 10 (as NH$_3$-N)

NH3-N/TKN fraction	Operation	Animals present			Kg NH$_3$-N/day
		Sows & boars	Finishers	Piglets	
0.62	Feed to finish		7,480		143
0.62	Total farm	1,212	7,480	1,410	211
1.00	Feed to finish		7,480		230
1.00	Total farm	1,212	7,480	1,410	340

5.1.2 Mass Balance Based on Dutch Model

The Dutch NH$_3$ emission methodology (see Section 4) is based on the average yearly nitrogen excretion per animal and the different emission or volatilization factors from specific emission sources; i.e., house, storage/treatment, and land application. The nitrogen excretion is the difference between the nitrogen that is ingested by the animal and the nitrogen that is ultimately bound in the agricultural product. There are numerous Dutch volatilization factors that have been determined for different animals, animal houses, waste storage/treatment systems, and land application situations (Staatscourant, 1998).[18] According to this source, the volatilization rates best matching a standard North Carolina swine house and lagoon are 15 and 15%, respectively.

The mass balance approach cannot be applied beyond the lagoon, because insufficient information exists about the amount and type of nitrogen that is stored in the sludge, escapes in the groundwater, or is emitted as nitrogen gas. When conducting a mass balance approach as illustrated here, the nitrogen content of the feed must be well known. According to Van Kempen (1999), the average feed for North Carolina finishing pigs has a nitrogen content of 2.99%; whereas, the average intake is 1.54 kg/day. If we assume an average animal presence of 96%, we can arrive at a total N intake of 16.16 kg N/animal/year. The average N excretion is 69%. Accordingly, the nitrogen mass balance for a pig at a theoretical feeder-to-finish farm is:

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[18] Compare to Federal Register.

- Input: 16.16 kg N per average animal present (AAP)/year
- N excretion: 69%
- Excretion: 11.15 kg N per AAP/year.
- Volatilization from house: 15%
- Emission: 1.7 kg N per AAP per year (or 2.1 kg NH_3 per AAP/year)
- N to lagoon: 9.5 kg N per AAP per year
- Volatilization from lagoon: 15%
- Emission from lagoon: 1.4 kg N per AAP per year (or 1.7 kg NH_3 per AAP/year).

The above emission factors are for finisher pigs. Sows and boars are about 3 times as heavy as average finisher pigs. (The piglets are ignored, because their reported weight is only 25 lb each.) Consequently, the emissions for Farm 10 based on the Dutch mass balance system are:

House (finishers): $2.1 \times 7{,}480 = 15{,}708$ kg NH_3 per year or 43 kg per day,

House (all pigs): $(2.1 \times 7{,}480) + (2.1 \times 3 \times 1{,}212) = 23{,}344$ kg NH_3 per year or 64 kg per day, and

Lagoon (all pigs): $(1.7 \times 7{,}480) + (1.7 \times 3 \times 1{,}212) = 18{,}897$ kg NH_3 per year or 52 kg per day.

The above calculation does not reflect the different feed that sows and boars typically receive, compared to finisher pigs. The 15% volatilization rate for houses, which was used here, is lower than most values found in the literature. (Some volatilization rates as reported in Section 2 are as high as 43%.) Furthermore, higher average temperatures in North Carolina, especially during the summer, may contribute to significantly higher emissions from the houses. Thus, the above emission factor for the house may be biased low. Also, the waste management system in the houses at Farm 10 is not truly representative of North Carolina practices. It is a flush-type system where the waste is assumed to be flushed to the lagoon every 8 hours (actually every 4 hours per half house). With a flush-type system, emissions from the house may be less (compared to a pull-plug system) because waste is removed more often. In a pull-plug system, waste is kept under a layer of water under the floor and flushed regularly; e.g., once a week. More frequent flushing of the houses would likely result in lower emissions from the houses but higher emissions from the lagoon and/or spray field. In addition, there are other more general limitations to the mass balance approach, which were discussed in Section 3.

5.2   EMISSIONS FROM HOUSES

Harris and Thompson (1998) reported an emission factor for finisher houses at Farm 10 of 3.69 kg NH_3/pig/year (see Section 3.1). This value may be compared to the emission factor of 4.31 kg/finisher pig/year which was reported by Harris (2001) who measured emissions at swine confinement buildings at other North Carolina farms. Applying both emission factors to the finisher pig population at Farm 10, results in NH_3 emission rates of 76 and 88 kg/day for Harris and Thompson (1998) and Harris (2001), respectively.

For comparison, the European Community emission factor for emissions from houses is 2.9 g per finishing pig per day (from Figure 3), resulting in a rate of 59 kg/day for Farm 10. Also, Groot Koerkamp (1998) includes emission factors for different houses (finishing pigs, slat floors) for four European countries (see Table 2). Paul (1997) estimated an emission factor of 4.6 to 7.0 mg N/hour/kg live weight for a swine house in Ontario, which also can be used to calculate a number for Farm 10 based on pig population and live weight. (When converting the Paul number, it is assumed that there are no diurnal effects on emissions.) Table 13 summarizes different estimates of emission rates standardized for finisher houses at Farm 10 based on emission factors from the aforementioned sources. The emission rates found by Harris and Thompson (1998) and Harris (2001) are somewhat higher than those found by the other North European and Canadian researchers. Most likely, this is due to differences in temperature, as well as in ventilation and waste management, and possibly feed.

Table 13: Ammonia Emission Rates for Finisher Houses at Farm 10

Researcher	Source	Emissions from houses (kg NH_3/day)
Harris & Thompson, 1998	Based on measurements in North Carolina	76
Harris, 2001		88
Van der Hoek, 1998		59
Groot Koerkamp et al., 1998 (English houses)		33
Groot Koerkamp et al., 1998 (Dutch houses)	Adopted from literature, based on European and Canadian emission factors	69
Groot Koerkamp et al., 1998 (Danish houses)		58
Groot Koerkamp et al., 1998 (German houses)		55
Paul, 1997 (Canadian house)		64 ± 13

5.3   EMISSIONS FROM LAGOONS

Over a period of a year, the lagoon at Farm 10 was sampled by three research groups. Table 14 presents the reported NH_3 emission rates for the lagoon at Farm 10, as well as the adjusted values for finishing pigs only. As mentioned, Farm 10 had 7,480 finisher pigs, 1,212 sows and boars, and 1,410 piglets. As mentioned, sows, boars, are about 3 times as heavy as average finisher pigs. If we assume that sows, boars, and finisher pigs produce the same amount of nitrogen in their waste, 33% of the nitrogen in the lagoon comes from the sows and boars. This is a simplification, because waste especially from sows may be more nitrogen-rich than waste from other pigs.

Table 14: Ammonia Emission Rates for the Lagoon at Farm 10

Researcher	Emissions from lagoon[a] (kg NH_3/day)	Emissions from lagoon, finishers only (kg NH_3/day)
Aneja et al., 2000	66.8	45
Harper & Sharpe, 1998	31.3	21
Todd, 1999	(165)[b]	No data available
Average Aneja and Harper & Sharpe	49	33

[a] See Table 8.
[b] Rejected (see below).

The value of 165 kg per day from Todd is rejected at this time, because it is greater than the entire farm mass balance calculated from the nitrogen excretion values. Similarly, it is greater than the sum of the house and lagoon estimates calculated by the mass balance methodology. Averaging the results from Aneja and Harper & Sharpe one can arrive at a lagoon emission factor of 49 kg per day for all pigs and 33 kg/day for finishers only. Per finisher, the NH_3 emission factor is 4.4 g/day or 1.6 kg/yr.

5.4   EMISSIONS FROM SPRAY FIELDS

No field tests on spray application were conducted as part of the studies at Farm 10. However, at a farm in Georgia, Sharpe and Harper (1997) used a micrometeorology method to determine emissions from a sprayed oats field of 12 hectares. Applied was 45 kg NH_3-N per hectare during three spraying events, 4.7 and 20.3 kg of which was volatilized during application and post-application, respectively. This translates into a volatilization factor of 56%.

The literature suggests a general NH_3 volatilization rate of about 50% in lagoon effluent (see Section 2.1), which is also in line with guidelines from the Dutch Mass Balance Methodology (see Table 11). This rate would include emissions during and after spraying. No distinction is made here between emissions from grass or crop land or tilled soil. Use of this rate requires lagoon effluent volumes and NH_3 concentration

data, but unfortunately, these parameters were not determined during the Farm 10 field tests. Some total N values for Farm 10 lagoon liquid exist, but no effluent/spray field flow rates were recorded.

We may approach the spray field emissions as the difference between the total Farm 10 NH_3 mass balance and the emission rates determined during the studies for the house and lagoon operations. Using the simple mass balance emissions rate for the finisher population of 143 kg/day, a house rate of 76 kg per day, and the lagoon rate of 33 kg per day, we arrive at a residue of: $143 - 76 - 33 = 34$ kg applied NH_3 per day or 12,410 kg/year. Assuming that the Georgia emission factor of 56% is representative of the Farm 10 situation, we can estimate spray field emissions at 19 kg/day or 6,950 kg per year. It must be recognized that spraying operations happen as relatively few events per year. As such, spraying events could be very significant during the actual spraying operations and the several days following.

5.5 DISCUSSION

Table 15 summarizes NH_3 emission rates from houses, lagoon, and spray field for Farm 10, as well as the overall rate from a mass balance approach. The emission rates for the houses in Table 15 calculated by various methods show good agreement and suggest that the houses are a more significant source than previously thought. The emission rate from the Dutch mass balance approach for finishing houses is lower than those of the field tests, but this may likely be due to the low volatilization percentage that was used in the mass balance computation. Consequently, it is believed that there is enough basis to recommend an emission factor for average finisher pigs for houses based on the Harris and Thompson (1998) number, which is 3.7 ± 1.0 kg NH_3/year/finisher pig (60 ± 10 g NH_3/kg live weight/year). This value is supported by the 4.3 kg NH_3/year/finisher pig reported by Harris (2001) for other farms. The emission factor does not account for differences in feed, ambient temperature due to seasons, house construction, or in-house waste management. Nor does it account for mortality and vacancy. The significance of each of these parameters may be a topic for research in the future. It is suggested that the live weight number be adopted for other pig categories, until more accurate field data become available for separate categories.

Table 15: Summary of Farm 10 Ammonia Emission Rates

From	Activity basis	Emissions (kg NH$_3$/day)	Method	Source
Entire farm	Finishers only	143	Simple mass balance	ARCADIS (this report)
	All pigs	211	Simple mass balance (scaled up from finishers only)	ARCADIS (this report)
Houses	Finishers only	43	Dutch mass balance	ARCADIS (this report)
	All pigs	64	Dutch mass balance (scaled up from finishers only)	ARCADIS (this report)
	Finishers only	76	OPFTIR field test	Harris & Thompson, 1998
	Finishers only	88	OPFTIR field test	Harris, 2001
	Generic pigs	59	Literature	Van der Hoek, 1998
	Finishers	33-69	Literature	Groot Koerkamp, 1998
	Generic pigs	64	Literature	Paul, 1997
Lagoon	All pigs	52	Dutch mass balance	ARCADIS (this report)
	All pigs	67	Flux chamber field test	Aneja, et al., 2000
	All pigs	31	Micromet field test	Harper & Sharpe, 1998
	All pigs	49	Average Aneja, et al. (2000) and Harper & Sharpe (1998)	
	Finishers only	33	Average. (Scaled down from all pigs)	
Spray fields	Finishers only	19	Simple mass balance	ARCADIS (this report)
	All pigs	28	Simple mass balance	ARCADIS (this report)

There is surprising similarity between the field test results for the lagoon (average 49 kg/day) with the number calculated by the mass balance method, which was 52 kg/day. Both numbers are for the total swine population. Using an average emission rate of 50 kg/day, the suggested annual emission factor for NH$_3$ emissions from a swine farm lagoon in North Carolina becomes 26 g/kg live weight/year. This lagoon emission factor does not take vacancy and mortality into account, nor does it address differences in lagoon characteristics, such as pH, or climatological factors, such as temperature, rain, and wind. Additional study of lagoons aimed at enhancing understanding of nitrogen pathways (e.g., to sludge or to N$_2$) will assist in further developing a comprehensive mass balance.

By applying the simple mass balance method, spray field emissions at Farm 10 were estimated at 19 kg/day or 6,950 kg per year. This reflects emissions from finishers (61.4 kg) only. As was indicated earlier, this number constitutes a rough guess. The calculations in this section suggest that spray field operations are a small but significant fraction of total farm emissions. But, since spraying is limited to certain seasons and certain hours of the day, it is likely that these spray operations are quite significant during the actual events.

(Spraying is prohibited in winter and is limited to a few hours per day during summer, because of labor constraints.) This fact is of importance when determining control technologies.

The total of emissions for finishing pigs from Farm 10 is 128 kg/day or 102 g NH_3/kg live weight/year, based on the emissions from houses (76 kg/day), lagoon (33 kg/day), and assumed spray application (19 kg/day). The 128 kg/day number compares well to the number established in Section 5.1.1 by the simple total mass balance (143 kg/day). Therefore, it can be concluded that a mass balance approach can be useful in estimating NH_3 emissions from swine farms, especially those that do not employ pull-plug waste flushing technology (see Section 4.1.5).

The average weight of the swine at Farm 10 is 69 kg. If we assume that this swine population reflects a self-sustaining population, in other words, is similar to the average swine population in North Carolina, we can arrive at an emission factor of 7 kg NH_3/animal/year (using the 102 g NH_3/kg live weight/year number). This emission factor is a generic emission factor mainly based on field data for two farms in North Carolina for houses and one farm for lagoons. The spray field component was calculated using a simple mass balance approach based on nitrogen feed intake. This emission factor is comparable to other generic emission factors from the literature (Table 16). The three European emission factors in Table 16 are all somewhat lower than the North Carolina emission factor. The difference may be a result of numerous factors, including but not limited to different animal waste handling practices (use of lagoons and flushing with lagoon water) and lower average ambient temperatures. If we take the Bouwman and Van der Hoek (1997) emission factor (5 kg/animal/year) as a lower boundary, we may possibly suggest a range for the North Carolina emission factor of ± 2 kg/animal/year.

Table 16: Comparison of Ammonia Emission Factors for Swine

	NH$_3$ Emission Factor (kg per animal per year)
Battye et al., 1994	9.21[a,b]
Asman, 2001	5.4
Bouwman & Van der Hoek, 1997[c]	4.8
Van der Hoek, 1998; EMEP/CORINAIR	6.39
This report, based on limited field tests and theoretical spray field emissions estimation.	7

[a]   This number is now apparently considered to be a misinterpretation. The initial swine number used in Battye was 8.5 instead of 5.4 and, as such, the value from Battye et al. (1994) is likely to be biased high (Asman, 2001, personal communication).

[b]   Decimals have been added for the purpose of tracking the source of the data and should not be construed as representing accuracy.

[c]   Calculated.

5.6   RECOMMENDATIONS FOR FURTHER RESEARCH

Recommendations for additional data collection are presented below. It must be emphasized that emission rates from the different sub-sources (house, waste storage, spray field) at one farm are interrelated. Changes in animal management (e.g., feed at the head of the chain) will affect emissions, not only in the house, but also from the lagoon and, ultimately, from the spray field. As such, field test campaigns should be inclusive of all data sources.

- As a first step, an inventory should be made of different houses and waste management practices for swine operations in the U.S. This inventory should have the same operational, spatial, and temporal detail as the animal categories collected by USDA, U.S. Statistical Abstracts, etc. The mass balance presented in this report can be important in identifying parameters that are likely to influence emissions and to what extent. It is important to include information on uncertainties in these parameters from the start.

- Because the emission estimates that are reflected in this report are from a specialized operation (farrow-to-finish with flush-type waste management), it is recommended that the complete series of measurements be repeated at a "standard" feeder-to-finish farm with a pull-plug system.

- If other animal categories (e.g.; turkeys or cattle) are of interest, the same should be done for these categories.

- In this report, spray field emissions were computed indirectly. It is of primary importance that valid

data be collected to characterize this remaining source of fugitive emissions. To provide a cohesive data set, it may be necessary that the barn and lagoon emissions be characterized during the same field study.

- A field test program involving the collection of data during and after field application is suggested. The development of a methodology that can relate the spray field emissions to animals present or live weight should be part of any field test program. The best approach to field-testing is to use scanning OP-FTIR with a vertical, 2-D optical plane downwind of a spraying operation. The acquired data represent the effluent plume as a sparse matrix. In combination with meteorological data, computerized tomography techniques are used to smooth the plume and calculate the emission rate. This experiment should be continued post-spraying. Required ancillary data for the generation of emission rate and its conversion to emission factor would include meteorological data, lagoon liquor ammoniated nitrogen concentration, time of spraying, pumping rate, cumulative animal population, and cumulative live weight. It is important to recognize that the lagoon represents the cumulative population rather than merely the current population. If available, a scanning tunable diode laser (TDL) spectrometer could be used in preference to the FTIR. A TDL offers faster operation.

- A relatively short and simple paper study could provide an independent check on the spray field calculations performed in this report, if additional pertinent data from the period of the North Carolina studies could be retrieved. Lagoon ammonium concentration, spraying events occurring during the study period, and spray volume(s) (data that must be maintained by farm management) could be used in concert with the literature volatilization rate of 56% to provide a "sanity" check on the difference calculations used in this report.

- Before field tests for spray fields are planned, a brief literature survey of older pre-1994 emissions studies from Europe and the U.S. could be useful. It is assumed that older European studies are available from before the time (around 1990) that spraying was discouraged in favor of low-emission technologies. This literature survey could be expanded to include the historical motivations to reduce spraying in favor of injection or other methods and the resulting emissions reductions.

- It may be worthwhile to conduct an experiment with two different tracers at a lagoon: one tracer that is dissolved in the lagoon water and another that is released in the air. In this way one could get an impression of the exchange process across the air/water interface as well as from the emissions that are released from lagoon water that is being used for flushing the house.

- As yet, not much thought appears to have been given to the fate of lagoons and lagoon sludge.

Nitrogen compound release as a result of lagoon sludge spreading is of interest.

- It might be worthwhile to consider which ecosystems are most susceptible to damage from direct NH_3 deposition. The NH_3 fluxes into and out of such an area can be measured using passive open-path Fourier transform infrared (OP-FTIR), possibly by making use of an airship. The approach suggested here applies airborne, down-looking, passive OP-FTIR to collect NH_3 path-integrated concentration data with the ground as the infrared radiation source. This will directly provide the plane-integrated concentration in a vertical cross-section beneath the route of the platform flight. Flying downwind from large NH_3 area sources like hog farms will allow the estimate of the total NH_3 emission flux.

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APPENDIX A. GENERAL ANIMAL FARMING AND ANIMAL WASTE MANAGEMENT PRACTICES IN THE U.S.

1.0   SWINE FARMING

A typical modern swine farm in the U.S. consists of a number of houses (anywhere from 2 to 10). Waste storage and treatment usually is by means of an anaerobic lagoon and a spray field. This setup is found throughout the Southeast and in other States where new farms are built. Older farms in the Midwest may have a concrete storage basin instead of a lagoon.

The pig houses are flanked by feeding bins, and there usually is a simple structure that serves as equipment storage and repair shop. Almost all farmers are under contract to a large company that provides pigs, feed, as well as general and environmental assistance. The farmer, usually does not live on the farm and works the farm alone or with one helper. For special jobs (e.g., cleaning and disinfection of the houses), contractors are used. Most farms are feeder-to-finish or farrow-to-finish operations; i.e. they receive pigs weighing 10 – 20 kg (22 – 44 lb) and keep them for about 180 days until they are about 115 kg (250 lb). Other farms specialize in pig breeding and are usually referred to as farrow-to-wean or farrowing operations. These farms have sows, piglets, and boars under one roof.

In North Carolina, there is a rather elaborate regulatory and assistance process, where the farmer is permitted or receives technical assistance from State and Federal government (e.g., the Natural Resources Conservation Service) and university Extension Programs. For example, lagoon design must be approved before a lagoon can be built. During construction and operation, there are regular inspections. Also, there must be a plan for land application of the waste that details application amounts related to the soil type and acreage.

1.1   Houses

The houses are approximately 70 m long and 15 m wide. They have 40 to 50 pens of an average size of 13 to 16 m^2 that contain 20 to 25 pigs each. The average space per pig is 0.6 to 0.7 m^2. There is a central aisle. The pens and aisles are separated by fences. The floors are made of concrete and are completely slatted, including under the central aisle. Waste produced by the pigs falls through the slats or is kicked down at a later stage. Under the entire house, there is a concrete compartment about 0.5 m deep with rough, unfinished sides and floor. This is filled with a layer of lagoon water of 25 cm (12 in.), which functions as a water seal

to reduce odors. The floor of the compartment gently slopes and, at the deep end, there are one or two plastic pipes or plugs that can be lifted manually to drain the pit. This is usually done once a week. The waste and water drain by gravity into the lagoon. This system is referred to as a "pull-plug" system. In rare occasions one may find houses with partially slatted floors and gutters under the slats that are flushed out by lagoon water intermittently. Pigs are intelligent creatures that would typically try to keep the non-slatted part of their pen clean.

On the back of the house (short side nearest the lagoon), five large automatic suction fans create a negative pressure in the house. These fans provide cooling in summer and remove dust, ammonia, and odor. Air inlets are either on the roof or on the other short side of the house. Also, some air may enter through the sides of the house, which are typically made of canvas/plastic. When the temperature gets above 32 °C (90 °F) the pigs are sprayed with a water mist, usually well water, to cool them. The spray comes on intermittently.

Feed is automatically distributed from silos next to the house by electric motors. Each house has two silos. Either pellets or granular feed is used. Both feed and water are continuously available to the pigs. Drinking water is dispensed by pressure-regulated spouts. Also, the houses are fitted with a movable ramp for loading the animals into and out of trucks. Between occupations, the houses are thoroughly cleaned by means of a power washer. They are also fitted with sprinkler systems for cleaning and disinfecting. Dead pigs are removed daily from the houses and are either buried or hauled off by a contractor for rendering.

1.2   Lagoons

The lagoon normally is an earthen basin with a clay liner. Some older lagoons were built before liners were required. A typical lagoon may be about 4.3 m deep, with the water level at around 3.3 m. Depending on the lay of the land and on the height of the water table, lagoons may be either dug or constructed with dikes. The size of the lagoon depends on the number of pigs and is typically designed to have a 180 day storage capacity and to have additional capacity to accommodate rain from a large storm. Actual retention time will vary with the land application requirements and the need to keep the water level in check. In other words, during wet weather, it will be necessary to spray more frequently to reduce the water level in the lagoon. A large farm may have more than one lagoon. The lagoon(s) can be classified as anaerobic, although there clearly is a facultative surface water layer that helps to control odors. Water for spray application is pumped from about 30 cm below the surface. After a number of years (e.g., 10 – 15 years), the lagoon will have to be

dredged to remove solids that have built up. Lagoon solids are expected to contain a significant amount of nitrogen.

1.3   Land Application

Typically the farmer grows grass for hay or a crop, such as corn or rye. Land application of lagoon effluent takes place from April through September, during dry weather in the mornings and afternoons. The effluent is sprayed by a large sprinkler through the air, several meters high and perhaps to a distance of 10 m. This device is referred to as a traveling gun. Typically, about 1.5 cm is applied per spray period which lasts 1 or 2 hours.

2.0   POULTRY RAISING

2.1   Houses

A poultry house typically may contain over 100,000 birds. Typical housing systems include solid floors with litter, or systems where the birds are kept in suspended cages above pits, where the manure is flushed out into a lagoon, or where it is kept in dry form inside the house (high rise or deep pit). In North Carolina, litter systems are, by far, the most common, followed by high rise pit systems. Pullets may be kept in cages with a pit before being moved to a house with litter. There is anecdotal evidence that odors from the area where pullets are kept are significantly stronger than from the litter system (Carter, Tom, NCSU, Extension Specialist, 27/4/99).

In the litter system, the birds are raised on concrete or earthen floors. A 2 to 6 in. layer of sawdust, wood shavings, rice hulls, or chopped straw is spread before the birds enter the building. Feeders and waterers hung from the ceiling can be raised or removed for cleaning. Litter management includes removing caked manure from around waterers, tilling to increase drying, and adding new litter. Periodically, tractor loaders remove the manure-litter mixture and pens are cleaned and disinfected. Or just the top layer of caked litter may be removed. The manure-litter mixture is land-spread or it may be stored for a few weeks to be land-spread later. This system is often used for turkeys, broilers, ducks, and small layer flocks.

In the deep pit system, cages for layers or pullets are in rows of decks suspended above a pit approximately 6 ft deep. The pit may be built in the ground or above ground. When the entire structure is built above ground, it is referred to as a high-rise house. Manure falls through the wire floor into the pit directly or is scraped from dropping boards below the cages. If kept dry, manure can accumulate in pits for at least a year

and often longer. Deep pits hold manure stored as a solid (typical for layers or pullets) or a liquid (typical for ducks).

The high rise layer house with deep pit storage is generally regarded as one of the most labor efficient systems for manure management. Its success, however, is very dependent on good drainage, prevention of water spillage onto the manure, and good air circulation over the manure mass. Drying should be enhanced by forced air circulation underneath the cages. Another factor affecting the manure moisture content in a high rise house is the bird density or number of birds per unit area of manure storage. Lower bird densities increase the coning effect of the manure thereby exposing more surface area to air flow and increasing drying. As bird densities increase, manure cones become less pronounced and are exposed to drying conditions for a lesser period of time before being covered by fresh manure. Manure is usually removed from the high rise house once per year. However, with proper moisture control, cleanouts have occurred as infrequently as once in 7 years (NCSU Extension webpage).

2.2   Manure Management Outside the Houses

Manure stored outside the building can be either liquid or solid. Properly designed and installed prefabricated steel, concrete, or earthen storages can be used for liquid poultry manure. With insufficient indoor solid manure storage, or with daily operation of scrapers in shallow pits, solid manure can be stacked outside the building. Increased storage can permit much longer intervals between land applications. Spreading is usually done in the spring or fall when land is most accessible. Plowing or soil incorporation soon after spreading is advised to conserve nutrients, to reduce field odors, flies, or other pest problems, and to prevent pollution from rainfall runoff.

The use of flushing gutter manure removal systems is a proven successful manure management technique, which involves moving large quantities of recycled lagoon liquid through the building on a daily basis. Advantages of the flush system include a clean in-house environment with positive control of gases and odors and an effective control of flies. Gutters are usually flushed daily with each gutter requiring about 15 minutes for cleaning. An outside collection box that couples with a smooth-walled drainpipe conducts the wastewater back to the lagoon. Lagoons must be sized properly to achieve odor control and a water quality suitable for flushing. North Carolina recommendations currently are 15 ft^3 of lagoon space per bird for caged layers for a single anaerobic lagoon. This system is rare in North Carolina. In view of pending legislation to phase out all animal waste lagoons, it is unlikely that new lagoons will be constructed (Carter, Tom, NCSU, Extension Specialist, 27/4/99).

3.0 DAIRY AND BEEF CATTLE RAISING

3.1 Dairy Cattle

There are two types of dairy cattle management, dry lot and open lot. The dry lot practice of dairy farming consists of a milking herd of dairy cattle that are confined to a lot or loafing area. The area is generally made of concrete and has three distinctive portions:

- First there is the feeding area. Whether feeding silage from a silo or pit, or feeding a total mix ration (TMR), the feeding is usually done in a trough with a conveyor system that distributes the feed evenly along the distance of the trough. Feed is available from either side of the trough. The manure from the feeding area is usually scraped from the area into the lagoon or pit with a skid loader or similar type of equipment. The feeding area is always covered.

- The second area is the loafing area. The loafing area is a covered shed with stalls on either side that are approximately 4 ft wide by 8 ft long, separated by a hall. Depending on the herd size, there can be two and sometime three hallways. The reason behind the short length is that the cattle can fit partially in the stall and ideally when the cow releases manure, it will be out in the hall and can be scraped into a holding pit or lagoon. The loafing area provides a low stress area to lounge and rest and usually has sawdust or wood shavings for bedding, sometimes sand. Often there are a number of fans and sprinkler systems to cool the cattle.

- The third area is the milking area. Here the person who is milking stands in a depressed area about 3 ft deep. Cattle are generally fed while being milked or shortly before or after milking. The cattle enter the area on both sides of the depressed part. The manure and urine are washed with water down drains to the lagoon or holding pit. Often there are amounts of iodine and other disinfectants that are used to sterilize the teats of the cow before and after milking.

The holding pit or lagoon is generally near the feeding and loafing area; e.g., around 50 ft away. The holding pit would be covered. The retention time of the manure depends on the size of the pit and the herd size. Water is used to clean the milk barn after milking, and the holding area that is used to hold the herd preceding milking. Sometimes, water may also be used to flush the hallways and other areas that need cleaning. In the dry lot practice, manure is most always scraped daily into a lagoon or holding pit. All concrete surfaces are curbed and slope toward the lagoon/pit. The lagoon is emptied as often as yearly or every 2 years for crop use. Before this occurs, the lagoon is "slurried" with water to soften solids that have settled and is spread onto crop or pasture land.

Problems that often occur with dry lot dairies are manure run-off, microbiological contamination, and odor. During high precipitation events, manure can wash off and over concrete areas, especially those that are in need of repair or those that are improperly designed. Coliform and other types of bacteria can be expected in dry lot manure. Footrot, pneumonia, and other bacteria related illnesses are common. Odor is a constant problem. Lime is often used to "dilute" the odors as well as good management practices. Flies, fly larvae, and other insects can be a problem also. The run-off is likely to contain large amounts of ammonia that is generated in the urine of the cattle. Urine that is allowed to dry on solid floors, slats, and other structures is probably the largest source of ammonia emissions.

In an open lot situation, the dairy cattle are kept in an open pasture or lot between milking periods. The lot is grass or earth and generally grazing is available. The cattle are brought into a feeding lot and wait to be milked, and then released. The cattle generally have access to a feed trough that has a concrete floor and is usually close to the milking parlor. The cattle eat going in and out from milking, and at their leisure. Manure management is the same as with dry lot, except the loafing area portion of the scheme is missing.

3.2   Beef Cattle

Beef cattle operations fall into three basic categories: cow-calf operation, stocker operation, and feedlot operation. The cow-calf operation is one in which brood cows are put to pasture, generally one cow-calf pair to every 3 acres. Cows have a calf every year; the gestation period for a cow is 285 days, and the cow is given approximately 60-90 days rest. Most calving is done either in the early spring or early fall. Calves are weaned at 7 months at a weight of 450-650 lb.

Problems encountered with a cow-calf operation include overgrazing, disease, and soil contamination from salt and mineral stations. Overgrazing results in erosion and retardation of growth in existing grasses. Disease, such as e-coli and brucellosis are common problems that can affect wildlife and existing herds downstream of contaminated farms. Soil contamination from salt and mineral boxes[19] is a common problem. Cattle will congregate around these areas and scatter the minerals on the ground. There is usually a depression made from the cattle standing around the area. Often the boxes are placed close to water, and the minerals are washed into nearby streams and ponds. The cattle will often drink after obtaining minerals and, therefore they are deposited into the water source. Most ammonia emissions may be expected from feeding

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[19]   A box is any structure in which minerals are placed. It could be a 55 gal. drum with a hole in it, a wooden box, or a concrete structure.

areas and from areas where cattle tend to congregate regularly, such as streams, shade trees, and mineral stations.

Stocker operations work similar to the cow-calf operation except that they are weaned calves that weigh from 450 to 800 lb. They are put on pasture and fed a high carbohydrate ration to gain weight and then shipped to a feedlot. The same problems are encountered as with a cow-calf operation. An additional problem would be that there is usually a feeding area in which the calves are fed from a trough where erosion and nutrient runoff can be observed. Occasionally, calves are fed from a truck that deposits feed directly on the ground. The same environmental problems would be expected.

Feedlot cattle are similar to dry lot dairy cattle. They are confined to pens, generally 100 head to a pen. The pens are uncovered. The cattle can be fed from 650 to 1500 lb, although ideal finished weight is about 1200 lb. The cattle are generally of the same weight, frame, and breed. The cattle are fed in a bunk feeder, which generally is along a road so a truck with a side discharge can fill the bunk. The bunk has a slotted bar structure so only the head can fit into the bunk and reach the food. The floor is usually earthen material. The majority of cattle feeding is done in the Midwest. There, the winters are cold and the ground is usually frozen during that time. But during other times of the year or if a persistent rain occurs, this area will be very messy and manure and feed nutrient runoff is a major problem in feedlot operations. There also is a tremendous ammonia problem, although often not recognized due to the open environment. Feedlot pens are cleaned yearly and resurfaced with fresh soil.

APPENDIX B. NITROUS OXIDE AND METHANE EMISSIONS FROM
ANIMAL WASTE MANAGEMENT

1.0   NITROUS OXIDE

Bouwman et al. (1995) assesses uncertainties in global nitrous oxide (N_2O) measurements by comparing variants of inventories with source estimates inferred from inverse modeling techniques. The paper concludes that the analysis presented does not reduce the uncertainty of annual N_2O emissions from individual sources. All sources include uncertainties, but uncertainties in emissions from soils and oceans, the largest global sources, have major implications for the zonal distributions. The paper does identify N_2O from animal excreta as a significant global source. This source was recently identified by Khalil and Rasmussen (1992), who estimated a global emission of 0.2 - 0.6 Tg N_2O-N/year. Bouwman et al. (1995) assumed that N_2O emission was 1% of the N in animal excreta. This estimate results in a global emission of 1.0 Tg N_2O -N/year.

The only field tests that were conducted in North Carolina to determine emissions from swine waste application were targeted toward N_2O. Experiments were conducted in April-May 1997 at fields planted with winter wheat fertilized with effluent from swine lagoons, as well as with individual constituents. N_2O fluxes were analyzed over several days and soil properties were assessed. N_2O emission fluxes increased directly after application and tapered off to pre-application values after about 5 days. The maximum flux that was measured was 4,000 μg N_2O-N per m^2 per hour (Whalen, 2000).

Moisture is an important parameter; it was found that emissions increased after rainfall. The soils continued to have high concentrations of NO_3 and NH_4^+ after the N_2O emissions had returned to normal. Whalen (2000) points out that lagoon effluent has 94% ammoniated nitrogen as opposed to liquid manure or sludge which have longer term effects, due to gradual decomposition (several months). The author further indicates that the results strongly suggest that most N_2O production in response to spray field fertilization results from denitrification. It is estimated that between 0.05 and 1.0% of the fertilizer is lost as N_2O directly after spraying. As indicated, more may be lost during ensuing rain events. The study does not give emission factors where N_2O emissions are directly related to the volume of waste sprayed.

In a second study, N_2O emissions and soil characteristics were analyzed during three spray events of lagoon effluent over a period of 24 days during August/September (Whalen et al., 2000). The nitrogen in the liquid

waste was >90% NH_4-N (thus inorganic). Nitrification and denitrification occurred almost entirely in the upper 20 cm of the soil. N_2O fluxes as high as 9,200 µg N_2O-N per m^2 per hour were observed directly after application, but emissions decreased after several days to pre-application levels. The authors note that there were poor correlations between soil physicochemical parameters and the N_2O fluxes. The total fertilizer N applied and N_2O-N emitted were 29.7 and 395 mg/m^2, respectively. The fractional loss corrected for background emissions was $1.4 \pm 1.1\%$. A value reported for mineral fertilizers from an extensive study by Bouman (1994 in Whalen et al., 2000) is 1.25%.

2.0   METHANE

Animal waste management is required when livestock or poultry are confined in large numbers. Animal waste from large operations is either managed as a solid (e.g., poultry litter) or it is diluted with water to form a slurry to facilitate removal and transportation to a storage or treatment system. Liquid animal waste treatment systems include lagoons or liquid slurry systems. Depending on the type of animal operation, waste may be managed as a solid or as a liquid. For example, waste from swine and dairy operations is often managed in lagoons or liquid slurry systems; whereas, waste from other animal categories, such as poultry and beef cattle, is not typically managed as a liquid (Safley et al., 1992). Manure that is excreted and left in the pasture and manure that is managed as a solid are also believed to emit some CH_4 (Safley et al., 1992). However, compared to emissions from the rumen, emissions from solid waste from cows are insignificant (Williams, 1993).

Lagoons are large basins that hold liquid waste for a period up to several months. The lagoons used for animal waste treatment are typically anaerobic, implying that they are relatively deep; i.e., more than 12 ft. Liquid slurry systems are generally characterized by large concrete-lined tanks (pits) built into the ground. In these systems, the diluted animal waste is stored for 6 or more months until it can be land-applied, which usually occurs in late fall after harvesting.

Although liquid slurry systems are believed to be anaerobic, they are not necessarily sources of significant CH_4 emissions. The animal waste slurry in such a system is believed to be too concentrated to accommodate significant populations of methanogenic bacteria. The solids concentration (wet basis) of swine manure in a slurry system is typically around 4 to 5%; whereas, the solids concentration in the influent of a swine manure lagoon is around 0.3%. Also, the pH in liquid slurry manure is generally too low for significant

methanogenesis to occur. Hence, the primary function of liquid slurry systems is storage, as opposed to treatment, and these systems can be ruled out as a significant source of CH_4 (Barker, 1997; Lorimor, 1997).

The choice of liquid animal waste management systems at swine or dairy operations is determined by type and size of operation, historical preference, and climate. For example, dairy cattle may be kept at pasture for grazing when not being milked at the milk parlor. Only the waste excreted at the milk parlor would be collected. Very small swine or dairy operations with few animals usually do not incorporate a waste management system. Older farms and small to mid-size farms would typically have a liquid slurry pit, as is the case in eastern Texas (Chasteen, 1997). Large, modern operations (more than 1,000 head), where the animals are permanently confined, have a comprehensive waste management system (i.e., either a slurry system or anaerobic lagoon, depending on the local climate). In colder climates, liquid slurry systems prevail because the temperature in a lagoon would be too cold for significant anaerobic degradation to occur during late fall, winter, and spring. Lorimor (1997) provides examples of states that would use either lagoons or liquid slurry systems: operations in Iowa use very few lagoons, and operations in Illinois, Minnesota, and Wisconsin use none; Missouri makes some use of lagoons; and states at latitudes south of Missouri primarily use lagoons. Swine operations in North Carolina almost solely make use of lagoons.

To identify the degree of lagoon utilization based on climate considerations for waste treatment at large, modern dairy and swine farms, the U.S. was divided into three climate categories. Category 1 includes states with a relatively warm climate; i.e., Arizona, Arkansas, California, Hawaii, Maryland, New Mexico, Oklahoma, Tennessee, Texas, and Virginia, as well as other southeastern states. Large, modern swine or dairy farms in Category 1 states would almost solely make use of lagoons. Category 2 includes Colorado, southern Illinois, Kansas, Kentucky, Missouri, Nevada, New Jersey, southern New York, Oregon, Pennsylvania, Utah, and West Virginia. In these states, lagoon usage at large modern swine and dairy farms is common. All other states fall in Category 3, which represents a climate with prolonged cold winters, where anaerobic lagoons are not feasible.

2.1   Methodology and Emission Estimates

The methodology to estimate CH_4 emissions from animal waste lagoons is represented by:

$$CH_4\,Emissions_i = \sum (P_{si} \cdot S_{si}) \cdot TAM_i \cdot M_i \cdot COD_i \cdot N \cdot EF \cdot 365 \cdot 10^{-12} \quad [Tg/yr]$$

in which:

- subscript i denotes animal type; e.g., swine or dairy cattle;
- subscript s denotes state;
- P_i = number of animals;
- S_i = fraction of waste from animal type i in state s that is treated in lagoons (%);
- TAM_i = typical animal mass in the U.S. (kg);
- M_i = manure and urine production (kg/kg TAM/day);
- COD_i = average COD in fresh manure and urine (g/kg);
- N = removal efficiency (%); and
- EF = emission factor = 0.3 ± 0.1 g CH_4/g $COD_{removed}$.

The number of animals (P) can be obtained from the U.S. Department of Agriculture, which publishes comprehensive statistics on the Internet. In 1996, the number of dairy cattle in the U.S. was estimated at 9.4 million. This estimate does not include milk cow replacement heifers, because these animals do not spend time at the milk parlor. The total number of swine in 1996 in the U.S. was 58 million. In June 1996, the State of North Carolina had 8.9 million swine and a negligible number of milk cows. In addition to North Carolina, important swine raising states are Iowa, Illinois, and Minnesota (see Table B1). The most important dairy cattle states are Wisconsin, California, New York, Pennsylvania, and Minnesota (see Table B2).

Values for the fraction of waste S from animal type i in each state are based on expert judgment and on the information provided by Barker (1997), Lorimor (1997), and Chasteen (1997). Safley et al. (1992) include values for S for the entire U.S., stating that 25% of swine manure and 10% of dairy cattle manure are managed in lagoons. In recent years, there has been a tendency toward larger swine and dairy farms where the animals are fully confined. Accordingly, it is believed that the S values used by Safley are now too low. Tables B1 and B2 include values for S for significant swine and/or dairy states.

**Table B1. Methane Emission Estimates from Anaerobic Swine Waste Treatment Lagoons
in the United States in 1996**

State	Climate Class	Population (million)	S^a (%)	CH_4 Emissions (Tg/yr)
IA	3	13,300,000	0	-
NC	1	8,900,000	90	0.42
MN	3	4,800,000	0	-
IL	3/2	4,700,000	30	0.07
IN	3	3,850,000	0	-
NE	3	3,800,000	0	-
MO	2	3,550,000	50	0.09
OH	3	1,700,000	0	-
KS	2	1,320,000	50	0.03
WI	3	1,200,000	0	-
OK	1	1,150,000	90	0.05
MI	3	1,100,000	0	-
PA	2	1,020,000	50	0.03
OTHERS	2	7,610,000	35	0.14
TOTAL		58,000,000	average 28%	0.8

[a] Fraction of waste that is treated in lagoons

**Table B2. Methane Emission Estimates from Anaerobic Dairy Cattle Waste Treatment Lagoons
in the United States in 1996**

State	Climate Class	Population (million)	S^a (%)	CH_4 Emissions (Tg/yr)
WI	3	1,475,000	0	-
CA	1	1,260,000	50	0.41
NY	3/2	700,000	20	0.09
PA	2	648,000	20	0.08
MN	3	600,000	0	-
TX	1	400,000	50	0.13
MI	3	328,000	0	-
OH	3	285,000	0	-
WA	3	264,000	0	-
IA	3	250,000	0	-
ID	3	245,000	0	-
MO	2	185,000	20	0.02
OTHERS		2,772,000	15	0.27
TOTAL		9,412,000	average 16%	1.0

[a] Fraction of waste that is treated in lagoons

According to Adler (1994), the typical animal mass (*TAM*) for dairy cattle is 550 kg; whereas, Barker (1997) uses 636 kg. It was decided to use the average of the two values: i.e., 593 kg. (Barker collected miscellaneous field data from over 50 different farms in North Carolina.) For swine, the average *TAM* is 61 kg (Adler, 1994; Barker, 1997; Safley et al., 1992). Barker (1997) provides manure production rates *M* for dairy cattle and swine. *M* for dairy cattle is 0.087 ± 0.013 g/day.kg and for swine *M* is 0.082 ± 0.025 g/day.kg. These figures are similar to the ones used in Adler (1994). Average *COD* in fresh swine manure and urine is 103 ± 39 g/kg. For dairy cattle, *COD* is 128 ± 27 g/kg.

In this study it is assumed that the animal waste does not undergo significant biodegradation with associated CH_4 emissions prior to reaching the lagoon. Also, not all animal waste COD that enters a lagoon will actually biodegrade and contribute to CH_4 emissions; i.e., the removal efficiency (*N*) is less than 100%. Adler (1994) expresses the removal efficiency as "Methane Conversion Factor" (MCF) which is defined as: the extent to which the maximum CH_4 producing capacity is realized for a given manure management system. According to this document, the MCF of an anaerobic animal waste lagoon $N = 90\%$ on a yearly basis. Undegraded COD may, in part, accumulate in the sludge (e.g., as living cell matter) or it may leave with the effluent. Lagoon effluent is land-applied (sprayed over fields), where remaining organics degrade aerobically or are taken up by vegetation. Table B3 summarizes the parameters used in the CH_4 emission calculations for swine and dairy cattle.

Table B3. Parameters Used To Calculate CH_4 Emissions for Swine and Dairy Cattle Waste Lagoons in the United States

Parameter	Symbol	Unit	Swine	Dairy
Typical animal mass	TAM	kg	61	593
Manure and urine production	M	kg/kg.day	0.085	0.087
COD	COD	g/kg	103	128
Removal efficiency	N	%	90	90
Emission factor	EF	g CH4/ g COD$_{removed}$	0.3	0.3

Tables B1 and B2 include CH_4 emission estimates from lagoons for swine and dairy cattle in the U.S., respectively. Methane emissions from swine waste lagoons in the U.S. are estimated at 0.8 Tg/year and from dairy cattle are 1.0 Tg/year. Total CH_4 emissions from animal waste lagoons in the U.S. are 1.8 Tg/year, which compares well with estimates by Safley et al. (1992). Safley estimated that total emissions from animal waste lagoons were 1.4 Tg/year. Also, Safley estimated that total emissions from dairy cattle were

1.0 Tg/year and from swine, 1.1 Tg/year. These estimates include emissions from other sources than anaerobic lagoons, such as liquid slurry systems and manure that is produced by animals at pasture.

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APPENDIX C. COMPARISON OF FIELD TEST METHODS TO MEASURE AIR EMISSIONS FROM SWINE LAGOONS

This appendix provides some notes on the technologies used by the several groups involved in the "Intensive Study" at Farm #NC10. These technologies include flux chambers (Aneja et al., 2000), micrometeorology (Harper and Sharpe, 1998), open-path Fourier Transform Infrared Spectroscopy (OP-FTIR) (Harris and Thompson, 1998) and computer-aided tomography OP-FTIR (CAT-OP-FTIR) (Todd, 1999). Also see Chapter 3 of this report, entitled "Field Tests in North Carolina." (All references are in Section 6 of the basic report.)

1.0   FLUX CHAMBERS

The flux chamber method uses a plastic chamber of defined dimensions to isolate a portion of the source under investigation. During use, a zero-grade (zero background for the effluent being tested) compressed air source delivers a known flow rate through the chamber and carries the diluted effluent being tested to an appropriate analytical technique. By measuring the effluent concentration in the diluted exit stream and having set the carrier gas flow rate, effluent mass per unit time is easily determined. In the case of the Intensive Site, ammonia was determined by chemiluminescence after conversion to nitric oxide. It is important to note that this is the one flux determination technique that does not require the determination of meteorology data. The flux chamber isolates the covered area from ambient wind conditions. The diluent gas flow provides a known and constant artificial breeze.

Three important assumptions must be recognized.

- The source acts as an infinite source for the effluent being determined. More precisely, the effluent being determined is entering the test area at least as fast as it is being carried away by the diluent gas flow. In the case of a waste lagoon, this seems like a reasonable assumption. Flux chambers have also been used for determining ammonia loss from field applications of liquefied hog waste. It is unclear that this assumption holds up in soil applications.

- The selected test area is equivalent to the source as a whole. In other words, testing at map points x_iy_j and x_ay_b will provide the same results. It should be noted that the CAT-OP-FTIR work was performed based upon exactly the opposite argument; i.e., localized variations in effluent rate do exist. While the authors of this report have not seen the presentations from the tomography group, it is our understanding that such variability was found during the Intensive Study work. Also, the flux chamber work presents an ammonia flux rate model in which the emission rate is related to factors that include lagoon water temperature. In a waste lagoon heated partially by solar influx and described as having "sloping sides," a variation in water temperature with location is probable.

- The gas velocity is not a parameter which influences the emission factor; whereas, the model presented by the micrometeorology report (Harper and Sharpe, 1998) states that it is. It is unlikely that that gas-phase ammonia concentration, influenced by gas velocity, is not a factor in the several equilibria between total ammonia nitrogen in the lagoon to gas-phase ammonia.

Flux chamber technology has the advantage of being a mature technology, which may be easily and dependably deployed to those sources to which it is applicable. This can be seen in the current report where data are provided on a seasonal basis for the entire year. Quality assurance measures are well defined. These include the diluent gas background concentration, the diluent gas flow rate, adsorption/desorption rates for the effluent and the flux chamber walls, sample conditioning equipment, and the analytical

technique itself. Freedom from the collection of meteorology data and use of a tracer gas may also be considered advantages. Finally, the authors of this report do not believe that flux chamber studies are applicable to all sources. The technology is well suited to stationary, well-mixed, infinite sources such as waste lagoons. It does not seem applicable to house studies, or the actual spraying phase of field applications.

2.0   MICROMETEOROLOGY

The micrometeorology technique uses a vertical array of wind speed and temperature sensors operated in parallel, with effluent sampling occurring in parallel. In use, this vertical array is located in the middle of the source. The effluent sampling and analysis technique is adjusted to suit the compounds currently being studied. In the case of airborne ammonia during the Intensive Study, this consisted of collecting 4-hour averaged samples in sulfuric acid containing impingers, followed by sample storage, transportation, and colorimetric analysis in the laboratory. When a predictive model is to be developed, additional data regarding the source will also be collected.

As in the flux chamber technique, there is an assumption that the selected test location is equivalent to the source as a whole. This assumption may be more valid here, since significant convective gas mixing should occur in the air above the lagoon. Yet, the basic premises of the CAT-OP-FTIR technique and its results do need to be addressed at some point.

It is interesting to note that the calculation of effluent flux density from the collected data is dependent upon effluent gas concentration, wind velocity, and the associated sampling heights.

Micrometeorology has the advantage of being a mature technology which may be easily and dependably deployed to those sources to which it is applicable. The group involved in the Intensive Study has used this equipment at a number of hog waste lagoons and at least one field application. Micrometeorology was apparently also used successfully in an Oklahoma field application study. Freedom from the use of a tracer gas may also be considered an advantage.

Quality assurance measures are well defined but do bring up issues. This is particularly true for everything associated with the sample collection and analysis (e.g., the ammonia determination by colorimetry). Six independent samples are collected every 4 hours. There must be, therefore, six gas sampling pumps that must be calibrated and checked. While the quality assurance steps are not reported in the available reports, it is likely that the samples must be stored cold, shipped back to the laboratory, and analyzed within some maximum holding period. Blank and synthetic samples should exist at both the laboratory and field level to check for contamination and losses. It is fortunate that so many independent samples were collected during each period since the loss of any one sample does not invalidate an entire sampling period.

Another issue associated with the collection of grab samples for ammonia analysis is the time resolution of the data. It is likely that the other data were collected at a fairly high sampling rate; a point per minute is often typical for computer-interfaced sensors. The ammonia data have a much lower sampling rate of one sample per 4-hour period. This results in the averaging of data collected at the higher sampling rates and a much lower time resolution to the flux density results. This also, of course, impacts the predictive modeling that is presented in the report.

Finally, it is not clear that micrometeorology studies are applicable to all sources. As the existing data demonstrate, it is well suited to stationary sources where the sensor/sampling tower may be located centrally. It does not seem applicable to house studies, for example.

3.0 TOMOGRAPHIC OPEN-PATH FOURIER TRANSFORM INFRARED SPECTROSCOPY (CAT-OP-FTIR)

This is a new technology, and the Intensive Study work is reported as the first large-scale field study using this technique. This technique requires two or more scanning OP-FTIRs and several retroreflectors. For the determination of emission rates, a tracer gas, non-reactive and without interference from ambient species or the effluent under study, was released from the middle of the sampling area. Setup consists of carefully locating the spectrometers and retroreflectors around the circumference of the sampling area in such a way that the combination of one spectrometer and the retroreflectors covers the entire sampling area and that each grid point is covered along significantly different vectors for the two spectrometers.

During operation, each spectrometer is scanned sequentially among the set of retroreflectors and determines path-averaged spectra. Post-collection, path-averaged concentrations are determined by comparison to reference spectra for the tracer gas and effluent compounds under study; then the data are manipulated by tomographic software to determine the 2-dimensional distribution of effluent concentration.

It is important to note that there is no indication that meteorology data are required for this method. It is certainly true that most, if not all, of the effluent plume must pass through the analytical plane. It is also true that there is nothing in the tracer gas ratio calculation that requires meteorology data. Meteorological data were collected during this study, however.

This technology would seem to have three, possibly four, advantages.

- The operational costs, as opposed to capital and setup costs, must be rather low. Operational costs consist mainly of liquid nitrogen for the detectors, tracer gas (though sulfur hexafluoride is very expensive), and electricity.

- There is a high time resolution to the data. In this study, they report a complete sampling of the lagoon every 2 minutes.

- The use of FTIR provides the potential for simultaneous, multicomponent analysis. This is a very real benefit when compared to the flux chambers and micrometeorology where an effluent specific analytical technique must be selected and integrated into the system.

- If meteorology data are not required for this method of determining emission rates, that would be an advantage it shares with the flux chamber technique.

This technology is very hardware intensive and has a complex setup procedure. Each spectrometer and retroreflector is located in a plane parallel to the sampling surface after classical survey techniques are performed.

It must be admitted that it is unclear what the determination of geographical concentration distribution offers to the calculation of emission rates from a source such as a swine farm lagoon. While current results may raise some quality assurance questions for the flux chamber and micrometeorological techniques, there is little to indicate that the concentration distribution is needed for the calculation of an emission rate.

4.0 OPEN-PATH FOURIER TRANSFORM INFRARED SPECTROSCOPY (OP-FTIR)

OP-FTIR utilizes a high-resolution spectrometer with a (measured) open path through which the sample passes. Depending upon the instrument configuration, it may be operated either single-pass (source and detector at opposing ends), or folded-pass (source and detector in a single box with a retroreflector at the opposite end). In use, the open sampling path is located downwind of the source and depends upon ambient air movement to move the plume through the analytical beam. Since ambient air movement is used for sample introduction to the instrument, the tracer gas technique is used for emission rate calculation.

Meteorological data are also collected. They are needed because this is the one method that operates downwind of the source, and wind direction must be monitored to ensure that the plume is moving through the analytical path. Additionally, relative humidity is monitored and used to calculate a water correction when needed.

OP-FTIR has several advantages for emission factor determination.

- OP-FTIR is very flexible. The instrument is located downwind of the source along with the colocated meteorological station. Being a technique that provides simultaneous, multicomponent analysis, it can be applied to new situations quickly and flexibly. For the Intensive Study, it is the only technique to collect data for both ammonia and methane. It was the only technique to move from the lagoon to the house exhausts.

- The technique provides high time resolution data. As operated for the Intensive Study, it provided concentration data with 1-minute resolution.

- OP-FTIR provides a sensitive analysis. Depending upon the analyte, quantification limits may be in the ppbv range.

- Each analyte is "over-determined." This means that each analyte will have several lines in the spectrum; each line provides an alternative for quantification at a different sensitivity or avoiding a spectral interference from some ambient component (such as water vapor).

- The technique can determine multiple gas-phase analytes simultaneously. Gases that cannot be determined by OP-FTIR include the mono-atomic and symmetric diatomic molecules.

OP-FTIR also has some disadvantages:

- Common ambient species, such as water and carbon dioxide, are also detected by FTIR and can interfere with the analysis. Water is a particular problem since, particularly in the East, it is present in high concentrations and has minor lines throughout the mid-IR (which is where these instruments operate). This is dealt with by including water in the quantification step and using relative humidity data for an independent analysis of it.

- The use of ambient air movements for sample introduction is also an issue since wind shifts can quickly move the entire plume partially or completely out of the sample beam. When this happens, data are lost and either one must wait for the wind to bring the plume back or one must move the instrument. The wind direction is monitored continually for this reason.

- Changing wind directions is one of the main reasons why this technique requires continual support by field personnel when operating. This increases operating costs.

- There is an assumption that sampling is being done at the appropriate height. High wind speed may "lay" the plume over and take it below the analytical beam. After quantification, the concentration data are matched up by time with the meteorological data and a point with high wind speed or an inappropriate wind direction for the instrument orientation is eliminated prior to emission rate calculation.

- Traditionally OP-FTIR is performed with a tracer gas release to calculate the emission rates. This is a prerequisite when dispersion modeling is used for flux rate calculations. The newest approaches, using multiple beam paths and computed tomography, avoid this. Selection of an appropriate tracer gas can be difficult, particularly when parallel studies are being performed by other groups. Some tracer gases, such as sulfur hexafluoride, are extremely expensive.

- The detector must operate at liquid nitrogen temperatures. Depending upon the detector's dewar size, this severely limits this technique's potential for collecting data around the clock.

- The selection of an I_o spectrum. The conversion of the collected spectral data proceeds from interferogram to intensity and finally to absorbance, which is the form needed for quantification. The conversion from intensity to absorbance is a ratio calculation using I_o, or background intensity spectrum. (It must be noted that this is separate from collection of an upwind background spectrum, which is used to determine ambient levels of the analytes being studied.) The problem comes in determining the proper method for collection, or calculation of this I_o spectrum. Because of the open-path configuration and continuous data collection, there is no appropriate time to collect a "real" I_o spectrum. Most groups, including this one and the CAT-OP-FTIR group, have settled on the calculation of a synthetic background spectrum from, typically, an average intensity data spectrum.